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Critical issues in estimating ILUC emissions

Outcomes of an expert consultation
9-10 November 2010, Ispra (Italy)

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The views expressed in this report are those of the experts who participated to the workshop only, and do not represent the opinion of the Commission

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LIST of ACRONYMS

AEZ	Agro-Ecological Zones
ARB	Air Resources Board
BLUM	Brazilian Land Use Model
CAP	Common Agricultural Policy
CARD	Center for Agricultural and Rural Development
CGE	Computable general equilibrium
CET	Constant Elasticity of Transformation
CIS	Commonwealth of Independent States
CLUE	Conversion of Land Use and its Effects Modelling Framework
COSIMO	Commodity Simulation Model
CRP	Conservation Reserve Programme
DDGS	Dried Distillers Grains with Solubles
DG AGRI	General of Agriculture
DG CLIMA	Directorate-General for Climate Action
DG ENER	Directorate-General for Energy
DG TRADE	Directorate-General for Trade
EC	European Commission
EMPA	Eidgenössische Materialprüfungs- und Forschungsanstalt
EPA	US Environmental Protection Agency
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FAPRI	Food and Agricultural Policy Research Institute
FASOM	Forest and Agricultural Sector Optimization Model
FAO	Food and Agriculture Organization of the United Nations
FQD	Fuel Quality Directive (Directive 2009/30/EC)
GAEZ	Global Agro-Ecological Zones
GHG	Greenhouse gas
GTAP	Global Trade Analysis Project
GWP	Global Warming Potential
ICCT	International Council on Clean Transportation
ICONE	Institute for International Trade Negotiations
IIASA	International Institute for Applied Systems Analysis
IFPRI	International Food Policy Research Institute
ILUC	Indirect land use change
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
JRC	European Commission Joint Research Centre
JRC IE	JRC Institute of Energy
JRC IES	JRC Institute for the Environment and Sustainability
JRC IPTS	JRC Institute for Prospective Technological Studies
LCA	Life Cycle Assessment
LCFS	Low Carbon Fuel Standard
LUC	(Direct) land use change
MIRAGE	Modelling International Relationships in Applied General
MODIS	Moderate Resolution Imaging Spectroradiometer
NPP	Net Primary Production
PBL	Netherlands Environmental Assessment Agency
RED	Renewable Energy Directive (Directive 2009/28/EC)
RFS	Renewable Fuel Standard (US EPA)
TPRL	Tropical Peat Research Laboratory

Summary

At the request of DG ENER and CLIMA, the JRC organised in November 2010 an expert consultation, grouping world-recognised academics and experts in the field on Indirect Land Use Change (ILUC) effects caused by increased use of biofuels. This consultation aimed at discussing the main uncertainties related to ILUC estimations and to answer to the questions addressed in the public consultation.

The two days discussions focused in particular on the following items:

1. Land use change and greenhouse gas emissions (methodologies, datasets and uncertainties to locate ILUC and calculate GHG emissions)
2. Agro-economic modelling and uncertainties
3. Policy options

The views expressed in this report are those of the experts who participated to the workshop only, and do not represent the opinion of the Commission

Land Use change emissions

Cropland allocation

In order to estimate ILUC emissions, models allocate increased crop area to different types of land. To guide this, models consider land price, maps of land suitability, and proximity to transport infrastructure and existing cultivation.

Other important criteria that should be included in models are proximity to processing plants, and the ease of establishing land ownership, but these are generally still not considered at present.

Most models allocate increases in cropped area to fallow or abandoned cropland and grasslands first, thus using up the buffer of land, which goes in and out of cultivation as prices vary. Since around 1990 the cropland area has decreased in the EU and much more so in CIS countries. If this land returns to use, the LUC emissions (due to foregone sequestration in re-growth of natural land cover) would be relatively low. However, whether this would actually happen depends on local causes. Some reclaimed wetlands were re-wetted and these are unlikely to return to production. Moreover, land is irreversibly lost to agriculture every year in favour of urbanisation, road construction etc.

In southern regions of EU15 the land was abandoned due to uneconomic yields; this could re-enter production if prices increase due to biofuels demand. But in CIS and New Member States state subsidies disappeared, labour costs increased and land ownership changed. Some of these structural factors can be resolved with time, but not specifically because of biofuels demand. Notably in Brazil, the protected forest area has recently increased, and some models already consider this. However, according to some models (e.g. LEITAP) the inclusion of protected areas has limited impact on land price so have little effect on the area of land conversion. Moreover, unless nearly all of the forest is protected, the main effect is merely to shift deforestation to other regions: even in Brazil protection only covers a quarter of the total remaining forest.

Which land cover is lost?

Forests contain higher carbon stock than other land covers, so the proportion of forest used for cropland expansion is critical to ILUC emissions. Satellite images from different years are often used to regionalize the proportions of different land cover being converted to cropland. However, the difficulties in distinguishing different land uses lead to remarkably high inconsistency between maps, and thus high uncertainty in regionalized estimates. Techniques are improving though. A recent study [Gibbs et al., 2010] reported that since 1990 in the tropics more than 80% of the crop expansion was into intact and disturbed forests. In deforestation hot spots, such as SE Asia, it is worthwhile to use high-quality data available only at local scale rather than global databases, since the difference may significantly affect the global emissions.

Fraction of palm on peat

Sources disagree about the fraction of existing oil palm plantations on peatland, possibly because of the different definitions of peatland. However the Tropical Peat Research Laboratory in Malaysia have indicated that the fraction for *recent* expansion was approximately 30% in Malaysia, whilst NGO-sponsored surveys

indicate the fraction is over 50% for both Indonesia and Malaysia.

Emissions per hectare

Once the type of land use displaced by cropland has been identified, the accompanying emissions are usually estimated using IPCC guidelines. Winrock International also provides convenient tables of land use change emissions which are widely used[1].

Emissions from peatland can be a large part of the total LUC emissions. Undrained tropical peat-land is a carbon sink, but for palm plantations it must be drained; then the peat decomposes. Results were presented indicating that, due to the high carbon loss even after the first 5 years since drainage, when carbon loss is greatest, the IPCC 2006 guidelines (table 5.6 in the guidelines) value of 73 $tCO_2ha^{-1}yr^{-1}$ should be considered as the minimum realistic value. Nevertheless, some studies have used inappropriately lower values than the IPCC documents.

Agro-economic modelling and uncertainties

Yield Increases

All models allow yields to be affected by price and time-trends, and some also relate the rate-of-yield increase to crop price[2]. Models such as IFPRI-MIRAGE relate yield increases to the level of inputs (all fertilizer, capital labour etc.), but calibrating these functions is often difficult. Although there is a valid correlation between changes in crop price and fertilizer application (e.g. for US maize) the correlation between crop price and yield is uncertain.

Increasing yields by primarily adding extra nitrogen fertilizers could increase GHG emissions per tonne of production particularly in already intensively fertilized areas. However, there was discussion that in some cases only a small part of the yield increase comes from additional nitrogen fertilizers. The FASOM model used by EPA for domestic agricultural modelling reports the emissions caused by more intensive management with respect to extra N application. Most models aggregate all of the chemical inputs, but the GHG emissions depend on the proportion of N in the fertilizer mix. Farmers can also add K and P, which can have a large yield increase, but result in much less emissions.

In developing countries most of the increase in crop production since 1961 came from area increase, but in developed countries it came mostly from yield increase. However, yield elasticities in some recent adjustments to GTAP lead to the opposite conclusion. FAPRI-EU wheat scenario predicts only 15% of extra supply from yield changes, whereas IFPRI-IMPACT predicts 70% from yield changes for wheat scenarios. GTAP has about 40% from yield gains.

Marginal Yield

Most models assume that the yield on the new cropland is close to the regional average yield for the same crop. However, the GTAP and IFPRI-MIRAGE models multiply the average yields by factors of 0.66 and 0.5 respectively. In Brazil IFPRI-MIRAGE assumed a factor of 0.75.

Statistical data for US counties shown by FAPRI indicate that yields in counties where cropland expansion increased between 2006 and 2009 indicate ratios of 0.95 for corn, 0.82 for soy and 1.25 for wheat, compared to yields in counties with no cropland expansion in the same period.

Regarding changes in use on the existing cropped area, models assume that if a low-yielding crop (e.g. barley) is replaced by a high-yielding crop such as wheat, the yield will jump from the regional average barley yield to the regional average wheat yield. An overestimation of the yield would underestimate the LUC for extra wheat production. However, there was disagreement whether crop displacement would affect the yields or not[3].

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[1]There is an error in these tables for re-growth of forest in EU: this is of little significance except when used to estimate foregone carbon sequestration in EU crop scenarios as for example in E4tech analysis, where it causes an underestimation by roughly a factor of 3.

[2]FAPRI-CARD and the 2011 version of IFPRI-MIRAGE.

[3] It should be also noted that, depending on relative market prices, a farmer may decide to grow a specialty crop like rye on land that could produce higher yielding wheat.

FAPRI showed that the expansion of corn area in the US between 2006 and 2009 was offset by reductions in other crop areas such as cotton, rye and canola.

Land abandoned in southern regions of the EU with low yields will still have lower than average yields when the land is re-utilized, particularly if the same management practices as before are adopted again. Although the yields on abandoned land in former Communist countries (including some New Member States) may be closer to national averages, these are much lower than EU averages. Even assuming that all of the land abandoned in the EU would achieve the national average yield, this would still be only approximately 65% of the EU average.

By-products

Models in the JRC comparison showed that by-products from the production of ethanol save 30-35% of total extra crop production, and for EU biodiesel 55-61%.This is significant as the by-products are often used for animal feed and will replace some of the existing land requirement for animal feed, thus reducing land use change. However it was claimed that some of the economic models may be underestimating the amount of land recovered by by-products if they do not perform a protein mass balance between Dried Distillers Grains with Solubles (DDGS) and the animal feeds that are displaced by DDGS.

Food reduction

All of the models in the JRC-IE comparison show a reduction in food and animal feed (for meat production) demand as crop prices rise due to biofuel mandates. These savings are highest for ethanol and generally occur in the models which report the least ILUC. If the food reduction effect is removed, then ILUC emissions from all of the models in the JRC modelling comparison increase significantly, so that their ILUC emissions outweigh the direct GHG savings for all the biofuels, with the possible exception of sugar-cane ethanol.

Pasture effects

If pasture land is lost to crops, one might expect pasture to be replaced at the expense of forest. Models based on GTAP account for this, but models that do not account for this may underestimate ILUC emissions. About 18% of the World's cropland is cropped twice a year or more, and this fraction could increase if crop prices rise (as shown in FAPRI projections), making an effect similar to price-induced yield increase in models. However, very few models include this.

Economic models tend to underestimate the long-term interconnectedness of world production and substitution between crops, because they are calibrated on short-term annual data.

Policy options[4]

The final discussions addressed policy issues, in particular:-

- Does the modelling provide a good basis for determining the significance of indirect land use change?
- Are the impacts significant?
- Can we differentiate between bioethanol/biodiesel, feedstocks, geographical areas and production methods?

The experts unanimously agreed that, even when uncertainties are high, there is strong evidence that the ILUC effect is significant and that this effect is crop-specific. The sustainability criteria in the Renewable Energy Directive (RED) and Fuel Quality Directive (FQD) limit direct land use change (LUC) but they are ineffective to avoid ILUC, and therefore additional policy measures are necessary.

The use of a factor which attributes a quantity of GHG emissions to crop-specific biofuels was the favourite option discussed, but it was also agreed that policies should incentivise good agricultural practices, land management C-mitigation strategies and intensification on pasture lands.

On the other hand, the experts agreed that the increase of the GHG threshold will have only a limited effect on ILUC reduction.

[4] The representative from the U.S. Environmental Protection Agency did not participate in the policy options discussion.

1. Background

The European Commission (EC) is debating internally how to address indirect land use change (ILUC) emissions in biofuels legislation. The Directives 2009/28/EC (Renewable Energy Directive) and 2009/30/EC (Fuel Quality Directive) contain provisions on monitoring and limiting the possible ILUC effects, but also give the Commission the task to further explore the issue and to review the greenhouse gas (GHG) impacts from ILUC, in order to establish the most appropriate mechanism for minimizing it.

Considering the great complexity of ILUC estimations, the Commission is therefore consulting on a wide basis, seeking advice on both the scale and characteristics of the problem, as well as, if the scale of the problem is significant enough, how it should be addressed.

To support to the Commission to prepare concrete policy options, it is thus necessary to discuss and answer to the following key questions:

1) Does the modelling provide a good basis for determining how significant indirect land use change resulting from the production of biofuels and bioliquids[5] is?
2) Are the impacts significant?
3) Can we differentiate between:
  a) bioethanol/biodiesel
  b) feedstocks
  c) geographical areas
  d) production methods (i.e. ILUC mitigation actions)?

The studies carried out by various Commission services[6] identified a number of relevant uncertainties that causes differences in the results projected by different models and GHG calculations methodologies. Therefore, following a public consultation on the above issues[7] and two meetings with stakeholders held in September and October in Brussels, the Commission's DG CLIMA and DG ENER asked the JRC to organize an expert consultation with worldwide recognized scientific experts to discuss the main uncertainties related to ILUC estimations.

The two days of discussions (held in Arona (Italy) on 9th and 10th of November) focused on the following items:

1. Land Use Change and GHG emissions:

- Methodologies: are the available studies adequate to determine how significant ILUC resulting from production of biofuels is?
- Data and uncertainties: are the available datasets adequate to calculate emissions from ILUC? Are all the relevant parameters effectively considered by different methodologies?

2. Agro-economic modelling and uncertainties: how does the range of existing estimates change the results?

3. What are the policy options?

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[5] All references to biofuels in this document also apply to bioliquids.
[6] JRC studies available at http://re.jrc.ec.europa.eu/bf-tp. Other Commission's studies in public consultation may be found at http://ec.europa.eu/energy/renewables/consultations/2010_10_31_iluc_and_biofuels_en.htm
[7] Results of the public consultation and background documents are available online at http://ec.europa.eu/energy/renewables/consultations/2010_10_31_iluc_and_biofuels_en.htm

2. Assessment of Land Use Changes

A difficult task in assessing ILUC effects is to identify the regions affected by land use change and the types of land/soils in these regions. This information is essential to calculate the related amount of GHG emissions. Different studies are using different methodologies to allocate the new cropland demand in future scenarios predicted by agro-economic models and to calculate the soil Carbon and above/belowground biomass emissions. The results of the calculations from all the different methodologies identified the following two important issues:

- GHG emissions per hectare of extra crop area strongly depend on the regions and on the ratio of biodiesel to bioethanol.
- GHG emissions are not only determined by the size of the area, but also the type of land converted is of significant importance.

The second point above is particularly relevant to account for the divergence of emissions calculated by different methodologies, and the following questions were identified as key topics for the expert discussions:

- What are the best criteria to drive spatial allocation of extra cropland from biofuels production, and what are the parameters defining the conversion of non-cropland to cropland?
- Which proportions of new cropland will go onto forest, shrubland and pasture? (historical data vs. modelled data)

2.1Cropland allocation

2.1.1 Available methodologies

In order to estimate ILUC emissions, models allocate increased crop area to different types of land. Several different approaches used to allocate the extra land demand were discussed at this workshop.

A) JRC "biophysical" approach.

This methodology has been recently published in JRC report no. 24483 by the JRC's IES and IE institutes [JRC 2010c].
This study incorporates the output from two global economic models (IFPRI-MIRAGE[8] and AGLINK-COSIMO (JRC, 2010a)) on land use change as input data to calculate the related GHG emissions. The study used the models regional economic results and spatially distributed the cropland area change on a finer scale within each region, using "land suitability" criteria and distance from existing cropland: ILUC scenarios were thus integrated with ancillary information using land suitability maps as prepared by IIASA/FAO to guide the expansion of extra land.
The original approach of this study is the development of a harmonised spatial dataset and advanced analysis methods for all aspects of estimating GHG emissions. Global cropland data derived from McGill and Wisconsin University's M3 data sets[9] were merged with more recent general land cover data from the classification of satellite imagery. Climate regions and ecological zone data were processed from interpolated weather station data, as provide by the WorldClim data.

[8] "*Global Trade and Environmental Impact Study of the EU Biofuels Mandate"* study carried out by the International Food Policy Institute (IFPRI) for the Directorate General for Trade (DG TRADE),
[9] http://www.geog.mcgill.ca/~nramankutty/Datasets/Datasets.html

In the spatial allocation process, cropland demands provided by economic models are processed using global raster layers with approximately 10 km grid spacing. The process is guided by the criteria of land suitability, derived from IIASA/FAO Global Agro-Ecological Zones (GAEZ) data, and the distance from existing cropland. Crops with an increase in demand are first allocated to existing cropland that is released by crops with a reduction in demand. The remaining demand is allocated to new agricultural land. Land cover maps derived from MODIS data for 2001 and 2004 are used to distribute the new cropland demand to non-cropland areas (e.g. forest) within the countries inside the regions. To make a cross-check of data, the JRC also used GLOBCOVER and Global Land Cover data from SAGE M3 database 2000[10] and substituted one for the other. Results of this cross-comparison did not show any significant differences.
The outcome of the spatial allocation process allows for the computation of changes in land use, which result in changes in soil carbon stock, Nitrous Oxide (N_2O) emissions (from mineralisation) and levels of carbon in the affected biomass (Figure 1).

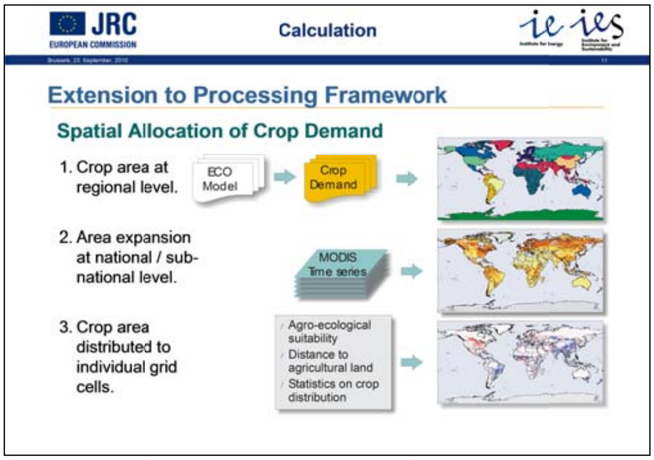

Figure 1: Schematic representation of JRC methodology to allocate extra cropland demand (slide presented during the expert consultation)

B) EPA approach

For the Renewable Fuel Standard–(RFS2) the US Environmental Protection Agency (EPA) developed a methodology to analyse the life cycle of renewable fuels, including indirect emissions, using Domestic Agricultural Sector Modelling, International Agricultural Sector Modelling and Land Use Change Modelling. The latter in particular uses long-term satellite data (MODIS: 2001-2007) to determine what types of land are impacted by international crop expansion. Domestic land use is modelled by the FASOM model, while international land use is modelled by FAPRI-CARD using Winrock International's data[11]. According to the Winrock approach, historical land conversion trends, as evaluated with satellite imagery, are used to determine what types of land are affected by agricultural land use changes in each country or sub-region.

C) GTAP based methodologies and IFPRI-MIRAGE CGE approach

With this method, the supply of land across different uses is determined through a Constant Elasticity of Transformation(CET) supply function. This CET function in GTAP is based on comparison of land rents between the three competing commercial uses: forestry, grazing and crops. Therefore, new cropland is allocated by estimating changes in the economic use of land (i.e. among forestry, cropland and pasture uses).
Although the IFPRI-MIRAGE-Biof[12] model uses part of the GTAP database (not related to biofuels or agricultural markets) the model is fully independent of the GTAP model. Whilst the GTAP model only includes 'managed' forest in its land use modelling the MIRAGE model also includes 'unmanaged' forest.

[10] http://www.sage.wisc.edu/mapsdatamodels.html

[11] Winrock International (2009) Winrock Land Use Change Emissions Results. Available at: http://www.regulations.gov/search/Regs/home.html#documentDetail?R=09000064809ad1c6

[12] MIRAGE-Biof is a specialized version of the MIRAGE model dedicated to the analysis of Biofuel policies and land use related policies.

The land use changes in the MIRAGE-Biof model are driven by a combination of mechanisms (listed below) that take place at the AEZ level (sub-regional level):

1. Among economic activities (crops, pasture, forestry/plantation) the land is allocated through a four level nested CET structure where land rents drive land allocation. In this approach, a key parameter is the value of elasticity of transformation used. At this stage, the MIRAGE-Biof model relies on different estimates coming from scientific literature. The value of elasticity explains how land is reallocated when relative rents evolve (the highest elasticity indicates the stronger the reallocation). The CET approach also implies that land reallocation leads to a change in average land productivity.
2. The evolution of land rent in agricultural activities drives land extension (or land contraction) through a land supply function. The elasticity of this land supply function depends on the region and is endogenous to the actual (and endogenous) ratio between total land use for agriculture and total land available and suitable for agriculture[13].
3. The previous stage defines the amount of land taken from (or added to) different ecosystems. To define which type of ecosystems is affected (forest, savannah, grassland etc.), the Winrock coefficients (EPA, 2010) are used. For both land extension and land reversion, the same coefficients are used. In both cases, the cropland extension ("deforestation") Winrock coefficients are used since it is considered that the time period used in the Winrock approach is not large enough to provide relevant shares of afforestation. National coefficients are used for all countries and AEZ except for Brazil where AEZ specific coefficients are used

   Due to the dynamic nature of the MIRAGE-Biof model and the reliance on national coefficients, it is possible that the availability of a particular type of land (e.g. Primary Forest) can become insufficient within one AEZ at one point in time to provide the amount of land needed, for instance:

   (Demand of New Land in period t in AEZ z) x Winrock Coefficient for type L > Available amount of land of type L in AEZ z in period t

   In such a case, a cross entropy approach is implemented to guarantee the physical constraint of available land.
4. At each period of time in the baseline and in the scenario, the amount of available suitable land for agriculture is corrected based on an historical trend. It aims to represent the demand for land taken (in most of the case) from agriculture for non-agriculture uses: urbanization, land protection program etc.

To summarize, the first and second effects are economically driven and endogenous to the model while the third and fourth effects are based on an historical approach.

In order to have sensitivity analysis, a closure of the model enable modification of the 1ststage (described above) to adopt an exogenous evolution (historically based) between pasture and crop lands. The previous description applies to the MIRAGE-Biof applications done in 2009 and 2010. New approaches are currently being tested (alternative functional forms and new calibration methods)

D) Life Cycle Analysis (LCA) method

A concept for modelling land use change in life cycle inventory has been developed by Schmidt et al. (2011). The model assumes that the current use of land reflects the current demand for land, and that land use changes are caused by changes in demand for land. The market for land is defined as a service that supplies

[13] If cropland rent increases by x % then total managed land (cropland, pasture, managed forest) will increase by y%, meaning that z ha are taken from the natural environment. y is endogenous and a function of x (the price change), country specific behaviour (price elasticity, that in this case will capture the way that land is managed and how the deforestation code is implemented for instance), current cropland area and total land suitable for agriculture. This is implemented at the AEZ level. Since current cropland evolves in the dynamic baseline and in the scenario, then Y is changing too.

capacity for production of biomass, i.e. the potential Net Primary Production (NPP_0). The starting point for the model is total global observed land use changes using a land use change matrix (See Figure 2).

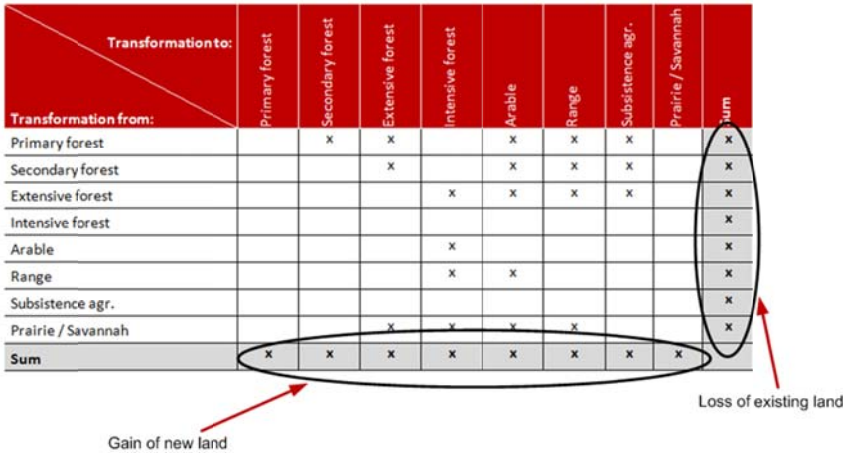

Figure 2 Land use change transition matrix (presented by J. Schmidt – 2.-0 LCA Consultants)

The link between LCA activities that occupy land and the land use changes is established via the markets for land. Indirect land use changes are modelled as upstream inputs to the product system. For example, Figure 3 shows an LCA process for wheat where the input of land tenure is the potential net primary production of the occupied land. This is used as the reference flow of the land tenure process that has four inputs: land already in use, expansion, intensification and crop displacement. The impacts (e.g. CO_2 from deforestation) caused by indirect land use changes are present within these four activities

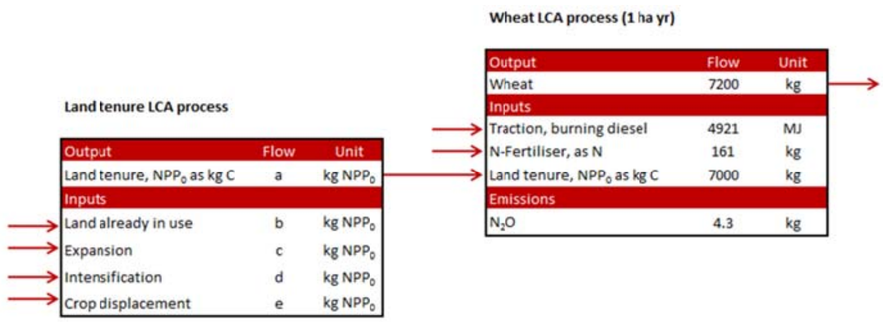

Figure 3 llustration of principle and mass balance of land tenure process

2.1.2 Discussion of the methodologies

Use of historical data

The importance of using historical data to project the conversion of natural vegetation to agricultural land was unanimously agreed. However, concerns were raised against some of the more commonly used historical databases.
Several studies rely on MODIS data (e.g. 2001-2004 or 2001-2007) to allocated marginal agricultural land expansion over different types of native vegetation. However, the patterns for land use change in different countries may not be entirely captured using MODIS data. Historic data, such as MODIS data, may not for example capture the recent trends in logging: some countries are implementing land management policies and deforestation reduction targets, and therefore deforestation in the future maybe very different from the past. It should be noted that this is a problem for the use of any historical data, not just MODIS data.

Another alternative method is the use of FAO LANDSAT analysis. However, also these data were shown to be inconsistent with reality in some cases. For example, particularly in developing countries, there might be significant differences (in some cases up to 30% according to IIASA) between FAO data and the real amount of new agricultural land available. Therefore, whilst FAO statistics may be correct for developed countries, errors in other countries may significantly affect the modelling exercise.

Methods based on Winrock approach

The Winrock approach is used not only by EPA in their RFS2 report, but in a number of other studies, as for example the work recently published by E4tech (2010)[14]. The Winrock approach applied the differencing method, also known as post-classification change detection, which involves comparison of two or more already processed land cover maps. Specifically, MODIS land cover maps (500m resolution) from 2001 and 2007 are subtracted to identify the land-use transitions during that time (2001-2004 at 1km spatial resolution were used in the first version). However, as noted by Winrock (2009) there are significant limitations to using global land cover products for relatively short observation periods to estimate land transitions. One issue is that the land cover classification uncertainty exceeds the rate of land cover change, which means that differencing maps can lead to spurious transitions. The table shown in Figure 4 presented during the discussion shows the unrealistically high percentage of change for most land cover classes. For example, 94% of the shrubland cover type was estimated to change to another land cover category over the 7-year period. Similarly, over half of the cropland pixels changed into other land cover types between 2000 and 2007. This shows that the differencing method with global data is unlikely to produce reliable results.

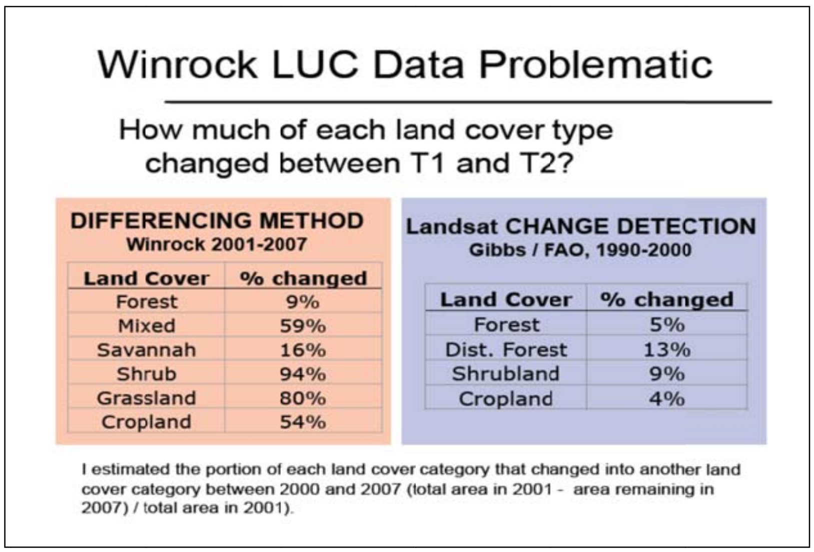

Figure 4 High percentage of change for most land cover classes [from H. Gibbs]

It was argued during the discussions that whilst the proportion of ecosystems converted to new cropland in Brazil reported by EPA in their RFS2 report using the Winrock approach is realistic (18% of cropland expansion onto forest and 35% on savannah), the amount of total area of cropland that changed in all of Brazil from 2001 to 2007 (31 Mha) is unrealistically high. This suggests that the Winrock data may be better at determining the types of ecosystems converted to new cropland than it is at measuring the total quantity of land use change during a given time period.

However, it is important to consider that in most of the cases the main problem is not the Winrock data itself, but how the data are interpreted and for what purpose they are used. It must be noted that these 2001-2004 data are showing “succession” occurring in just 2 to 3 years and are not showing what the final land use will be. Therefore, an incorrect use of the numbers might convey the wrong conclusions regarding the total C emissions or sequestration.

[14] E4tech is a business consultancy based in the UK and worked closely with the biofuel industry in producing the E4tech (2010) report: *A causal descriptive approach to modelling the GHG emissions associated with the indirect land use change impacts of biofuels*

Economic projections relying on CET functions

GTAP-based models don't include unmanaged land, (they only consider MANAGED forest converted to cropland), and the CET function used is based on comparing land rents. This is because, within a CET framework the new land has to come from an economically valued source (an economic trade-off between managed forest and cropland). Therefore, about 40% of the World's forest is not included, because it is unmanaged and has thus no rent. This not only gives an underestimation of the potential for forest conversion, but also prevents the capture of the whole economic pathway leaving out large sources of forest land. Timber products in GTAP for example can only be provided by converting agricultural land back into forest, because any new forest cannot be cleared. Moreover, CET functions are empirically derived, but because all the data are based on recent US experiences, and extrapolating economically driven LUC from CET functions for other countries might then be questionable.

2.1.3 Drivers for extra land allocation

To guide the allocation process models generally consider land price, maps of land suitability, and proximity to transport infrastructure, existing cultivation and processing plants.

When discussing the capability of present methodologies in capturing land use changes and correctly allocating the changes, it is not relevant to identify what is driving the demand (e.g. whether it be biofuels or soybean consumption in China or beef consumption in Europe). The key question is what is really driving the allocation of cropland changes. Land suitability is certainly one driver (it is one of the criteria used in the JRC methodology (JRC, 2010c)), but other socio-economic criteria and political constraints on land use in specific countries are also important. These criteria are generally not included in models; particularly in purely biophysical models (like CLUE[15] for the EU) that fail to account for political constraints on land use.

In Indonesia for example, land suitability, socio-economic criteria and proximity to existing cropland are not always the main criteria driving LUC. What matters more are the administrative boundaries, how easy it is to obtain the land for cultivation and the ownership (the land with the least ownership claims can often be obtained more easily).However, in most cases from an environmental perspective this also corresponds to the least suitable land, since in Indonesia this often happens to be land on peat. In cases like this, the use of good historical data may be able to more accurately determine the most likely areas for future land conversion.

2.1.4 Exclusion of protected areas from available areas

In some regions governmental programmes are increasing native protected areas: It was reported that in Brazil at present there are 108 Mha of native protection areas and indigenous reserves in the Amazon (out of a total land area of 400 Mha). The Brazilian Government is also setting up national parks in the Cerrado region, thereby reducing deforestation in this area by 40%.

It was indicated by PBL that when the LEITAP model was run for a global increase in protected areas of between 17 and 20 % there was little effect on the land prices in land abundant regions such as Brazil (thus a weak effect on the land conversion). LEITAP has regional land markets, and in reality the effect may be larger because the areas protected may be the most likely to be converted.

The most likely effect of increasing protection of forests will be a shift of deforestation to other regions. For example protecting only 20% of Amazonia in Brazil will move the deforestation elsewhere. It was shown that a similar shift has already occurred in Indonesia, where a 15% increase of protected forest shifted deforestation to Myanmar and Cambodia.

Some economic models are already excluding protected areas, for example: FAPRI excludes protected areas from agricultural land availability in its Brazilian module. LEITAP takes into account current protected areas using data from International Union for Conservation of Nature (IUCN).

No protected area constraint is included in IFPRI-MIRAGE, but IPFRI agreed that if this constraint were to be introduced in a CGE model the land availability would be reduced. In some regions there may be sufficient land

[15] CLUE (Conversion of Land Use and its Effects Modelling Framework) is a dynamic, multi-scale land use and land cover change model. http://www.cluemodel.nl/

available for expansion, but in specific areas this constraint will be activated. Moreover, the increased food demand in 2020 is pushing cropland into many regions, so marginally nature protection will play a role.

It was pointed out that whilst exclusion of nature protection areas is be applicable for models that are trying to calculate land suitability for the conversion of cropland (e.g. LEITAP or the JRC methodology) this is not the case for the economic models that use a US based CET function for controlling all world land use change.

It is important to know the C stock of protected areas as this can significantly affect the ILUC emission estimates.

2.1.5 Land conversion

As discussed above a key issue in estimating the emissions from ILUC is to identify the type of land converted and the proportion of new cropland that will go onto forest (and in consequence onto shrubland and pasture). Forests contain higher carbon stock than other land covers, so the proportion of forest used for cropland expansion is critical to ILUC emissions. Satellite images from different years are often used to regionalize the proportions of different land cover being converted to cropland. However, the difficulties in distinguishing different land uses lead to remarkably high inconsistency between maps, and thus high uncertainty in regionalized estimates.

Fraction of cropland on forest in the tropical areas

A recent publication from Holly Gibbs et al. (2010) was presented by JRC co-authors of the paper. In this study, FAO data from 120 sample sites (published by FAO in the Forest Resource Assessment for 2000) were subjected to complementary analysis of their satellite imagery with a higher resolution (30x30m). Land cover types produced by FAO for these images also included categories relative to permanent/smallholdings and fallow agriculture, therefore providing information about the use of deforested areas. Results of this study (summarized in Figure 5) showed that in the sampled areas (for the tropics) historically between the 1980 and 2000 more than 55% of new agricultural land increase is created from intact forests, and another 28% came from disturbed forests. The FAO results are also confirmed by the analysis of the Landsat database developed by the JRC Tropical Resources and Environment monitoring by Satellites project (TREES).

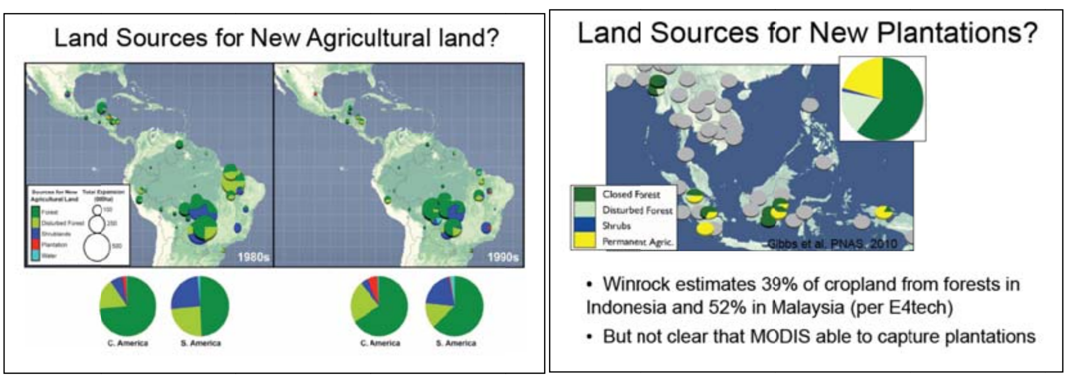

Figure 5 Sampled areas in the tropics (from Gibbs et al., 2010)

Some of the main limitations of the study are that it covers only about 10% of forested areas in the tropics[16], cropland and pasture classes are aggregated, and the data are limited to the 1980s and 1990s. But with no viable economic tools for predicting where the new cropland is going to come from, historical data remains the only valid alternative to have 'realistic' indications about the trends, although deviations from historical behaviour cannot be excluded.

Another problem related to the identification of the amount of forest converted to cropland (and which causes underestimations of forest conversion) is that in some cases primary forests have been deliberately

[16] FAO and the JRC are jointly working with a new bigger sample of land beyond tropics, covering also the rest of the world and the results will be ready in 2011,

reclassified as secondary (managed) forests, thus becoming 'suitable' for the cultivation of specific crops (e.g. Oil palms).

Fraction of cropland on forest in the EU

According to modelled results, part of the increase in land use in the EU will still come from "deforestation". However, it was argued that the increase of demand for EU biofuels crops may be met by a reduction in the rate of abandonment of arable land in the EU (i.e. land will be recovered from abandoned land) [17]. If this is the case, then the loss of carbon stock will be the carbon that would have accumulated on the idle land.
It should be stressed that, at least for the GTAP based models, "deforestation" only refers to managed forests being converted to cropland. This is because these models only include commercial forest plantations.
When making a shock on 1st generation biofuels, cropland prices will increase compared to the value of forest plantations, and therefore cropland will be taken from plantations or from a reduced increase of forest. In fact, in the IFPRI-MIRAGE baseline for the EU, expansion of forest plantations are expected , so this does not necessarily mean cutting existing forest.

Fraction of palm on peat

Undrained peatlands are a significant carbon sink but to support palm oil production peatlands must be thoroughly drained, with the resulting impacts: drainage lowers the water table, drying and thus decomposing, through oxidation, the peat that then becomes a carbon source (Figure 6).

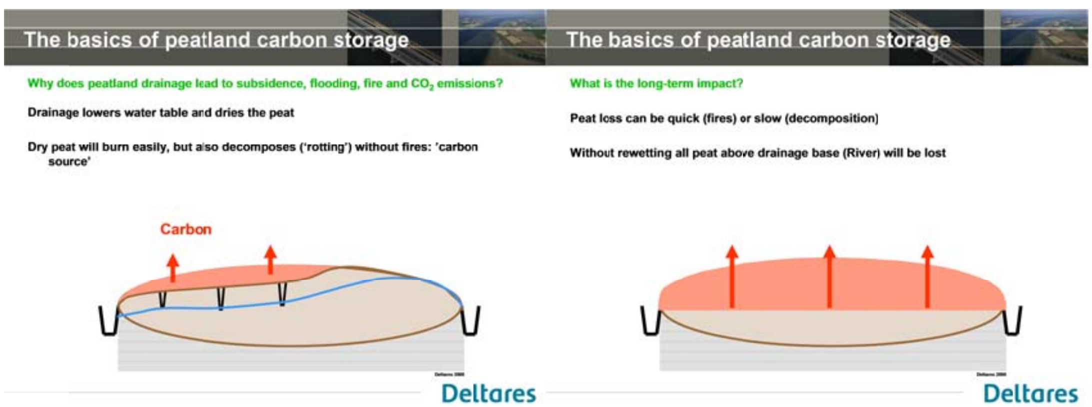

Figure 6: carbon emissions from peatland drainage [presented by A. Hooijer, Deltares]

It was argued by the Tropical Peat Research Laboratory (TPRL)[18] representative from Malaysia that the drainage accompanied by water control (compaction and water management) would increase the bulk density, thus improving on the capillary rise while also increase the water filled pore space for good plant growth. With compaction and water control, the moisture content of the peat increases and the rate of decomposition and peat oxidation actually decrease. This is certainly the ideal situation, but at present none of the available scientific studies give evidence of this practice. Moreover, data presented by other experts show that the increase in bulk density mostly occurs in the first five years after drainage, and is minimal in later years. This leads to the conclusion that the subsidence rate of around 5 cm yr^{-1} that still occurs after the first five years is caused almost entirely by peat oxidation not compaction, and therefore correlates directly with carbon emissions. It was argued that this dominance of oxidation as a cause of subsidence in tropical peatlands, compared to conditions in temperate peatlands, is due the difference in temperature of nearly

[17] In this case, to calculate the GHG balance of using abandoned land, one should also consider that **if** afforestation is going onto abandoned land, bringing this land back to cultivation for crops will lead to a loss of foregone carbon sequestration

[18] The Tropical Peat Research Laboratory is a research division established by the Sarawak government and funded by the Malaysian Palm Oil Board and other E-science fundings.

20°C. Studies in the Everglades (USA) have shown that the rate of peat oxidation doubles with every 10°C in soil temperature, at an even water depth.

The main regions affected by peat conversions are Indonesia and Malaysia, and on the basis of C storage Indonesian peatlands are likely to contain 84% of the entire C in tropical peatlands in SE Asia and 65% of the C in global tropical peatlands, estimated at 89 billion tonnes (not including the biomass of 69 Billion tonnes) (See Figure 7).

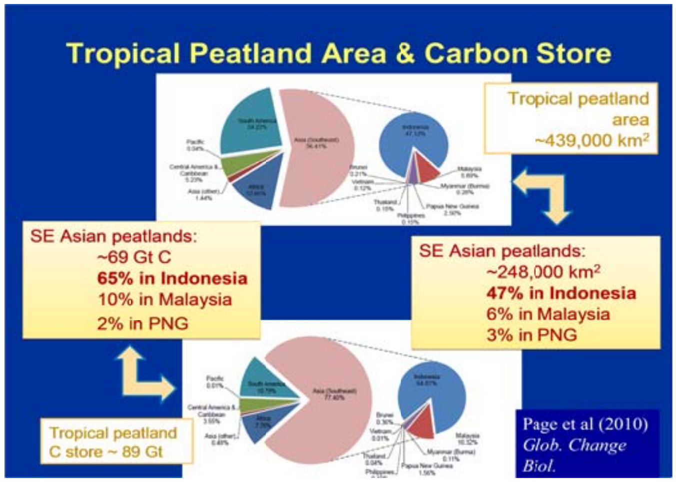

Figure 7: global distribution of peatland areas and C store [presented by S. Page – Univ. of Leicester]

Although it is evident that a large fraction of oil palm and agricultural area moved onto peat swamp forest (Figure 8), sources disagree on the exact fraction of existing oil palms that is on peatland. In some cases this is because of the different definitions of "peatland".

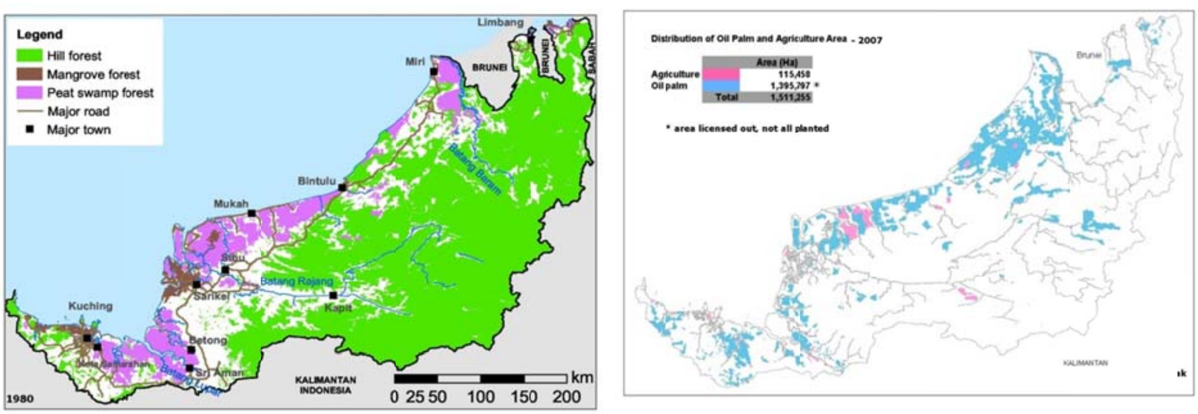

Figure 8: comparison of 1980 and 2007 land use maps for Sarawak region (Malaysia) [Source: Forestry Department of Sarawak[19] presented by H.J.Stibig – JRC)

In Indonesia at present about 5.5-6 Mha of peatland are estimated to be used for oil palm. Projections show that by 2025 oil palm and pulp wood plantations will cover more than 50% (10 Mha) of peatlands in Indonesia (unofficial logging may cause additional significant drainage).FAOSTAT data for Malaysia shows that in 2003 oil palm covered 0.31 million ha (9.5%) of total peatland area which rose to 0.51 million ha (13%) in 2008 ("TPRL, 2009" showed that land in Malaysia used for oil palm increased from 0.2 million ha in 2003 to 0.6 million ha in 2007). The TPRL representative claimed that the fraction of peatland used for oil palm plantations in Malaysia and Indonesia is lower than those reported (respectively 5% in Malaysia and 15% in Indonesia).

[19] Wong, J., Presentation made by the Forestry Department of Sarawak, JRC-MPOC workshop on *"Direct and indirect impact of biofuels policies on tropical deforestation in Malaysia", Kuala Lumpur, 20-22.Nov.2008*

3. GHG emissions from Land Use Changes

This section focuses on whether the available datasets are adequate to calculate ILUC emissions and if all the relevant parameters are effectively considered by the different methodologies.

3.1 Methodologies to calculate GHG emissions

The IPCC methodology is the most commonly used method to calculate GHG emissions due to LUC.

A complementary approach consistent with the method used for the assessment of biodiversity in LCA was also discussed. The Global Warming Potential (GWP) index proposed by the IPCC for assessing the contribution of different GHGs to climate change was not designed for application in LCA and is based on subjective time preferences. Instead, the approach of Müller-Wenk and Brandão (2010)(presented by J. Reinhard from EMPA) is consistent with the GWP index and with the method used for the assessment of biodiversity in LCA.
This method, based on Moura-Costa (2000), derives an equivalence factor between tonne C-eq. and tonne C-year. This factor is applied to biogenic carbon emissions from land and assessed against their fossil counterparts. The difference results from the different residence times in the atmosphere that arises. When land-use change (LUC) takes place, the carbon stored in soils and biomass is released. By doing so, an additional sink is concurrently opened on that land which is able to retrieve back the carbon emitted. In consequence, one tonne of C from LUC has a different impact than one tonne of carbon from fossil fuel combustion. The advantage of applying this methodology to LCA is that it considers not only carbon quantities transmitted to the atmosphere but also the mean duration time of these carbon quantities in the atmosphere. Furthermore, it is in line with the current LCA methodology to assess biodiversity impacts.
Some criticisms were raised on the possibility of the LCA methodology in particular, double counting the C sinks: it was in fact argued that the LCA method assumes that the cropland re-grows, and that the re-growth of a forest counts as a C sink. If the land is converted to cropland then the C should count the same or if the land is converted to cropland and the model assumes that there is re-growth of forest elsewhere (which happens in the GTAP model) then the C will be counted as a sink separately. If you have already accounted for that, you will be double counting. Most of the LUC models (e.g. IFPRI-MIRAGE) assume that the land use change is permanently converted to cropland. The IFPRI-MIRAGE model looks at the amount of land by activities (e.g. cropland managed forest etc.) and assumes a C value for each of these land types. (In the report by IFPRI (2010), it was assumed that in the EU the forest lost was new forest (from afforestation) which has half the C of a mature forest).
However, the authors of the discussed methodology argued that even considering land-use change as irreversible (quite a strong assumption), the impacts of land use (i.e. land occupation) are attributed to the current crop and not to the crops that originally triggered the land-use change. So, if it is only the triggering crop (i.e. biofuels) that you're assessing, you can only ascribe it the transformation, but not the 'unknown' subsequent occupation impacts

Time: Amortization over 20 years

In all cases results depend on the amortization period (the EU legislation decided to go for 20 years as specified under the Tier 1 approach of IPCC, while US legislation goes for 30 years).
It was argued that for policy comparison an appropriate discount factor must be chosen. It is important to understand that the time profile of LUC emissions (which are added to the direct emissions from biofuel use) is different from the time profile of the same amount of C emitted by fossil fuels. There is no discount factor in the LCA methodology and the claim is that there is no difference between a C discharge now and 500 years from now. In response it was noted out that the (Publicly Available Specification) PAS 2050 British standard (2008) does account for delayed emissions.

An understanding of the change in the long term situation beyond 2020 is also important for the sustainability criteria of biofuel production.

3.2 Accounting for foregone C sequestration

Proper estimations of ILUC emissions must also account for carbon sequestration, which has proven to be difficult to accurately assess. Most of the studies which estimated low ILUC emissions by allocating large amounts of cropland to abandoned land for example did not properly account for the foregone C sequestration when the land is cultivated again. The main issues identified during the discussions are summarized below.

1. Carbon sequestration due to afforestation

The rate of carbon accumulation on abandoned arable land is entirely different from the amortised carbon loss from deforested land and grassland. It depends on whether land is left idle (natural succession) or is afforested.

According to FAO data, 1,134,000 ha of land has been abandoned between 1995 and 2005 in the EU, of which 721,000 ha has been reforested. The E4tech (2010) study presented at the meeting applies afforestation gains to only 12% of these abandoned lands (138,000 ha), and calculates Soil C stocks on 88% of this land, without considering the amount of afforestation on active cropland. It was argued at the meeting that this underestimates the area afforested and hence foregone C sequestration.

The table shown in Figure 9 gives the range of afforestation and the rate of C accumulation associated with foregone sequestration in the EU that are assumed by different models.

Carbon stock change for converted cropland in the EU

Model	Method	% of new cropland from forest	C stock change t C/ha/yr	Source
Analysis of historic data	FAO	0%	0.48	Lywood 2010
E4tech reversion	MODIS	45%	1.5	E4tech 2010
E4tech conversion	MODIS	6%	1.7	E4tech 2010
IFPRI	Cost curve	34%	1.9	IFPRI 2010
JRC new methodology	AEZ etc	38%	1.7	JRC 2010c
GTAP	CET	77%	3.4	JRC 2010b

Figure 9: Afforestation rates in the EU assumed by different models [presented by W. Lywood – Ensus].

However, it was pointed out that some studies (e.g. E4tech, 2010) do not take into account belowground biomass in estimating GHG Emissions. .

For example, the accumulation rate of 0.48 t C ha^{-1} yr^{-1} reported in the figure above is just soil C from natural regeneration. Therefore, it is necessary to add vegetative carbon (~1.75 t C ha^{-1} yr^{-1}.) thus bringing the total closer to 1.5- 2 t C ha^{-1} yr^{-1}, while for plantation forests the C accumulation figure is closer to 2.5 to 3.5 t C ha^{-1} yr^{-1} [20].

2. Misinterpretation of Winrock reversion factors (afforestation)

In addition to estimating emission factors, Winrock International also developed "reversion factors" to estimate the carbon accumulation in biomass and soils that occurs when managed cropland and pasture land is abandoned, looking at re-growing forests and forest accumulation rates. However, the correct use and interpretation of these reversion factors has been proven to be critical for a number of reasons:

- The calculation of potential losses of C sequestration is largely underestimated if average sequestration of mature forest is used, and not the re-growth of freshly planted forest (as should be the case in EU for abandoned land).

- Winrock identifies the new use of land in 2006 that was cropland in 2001. This time period is not sufficient for forest to form a significant canopy cover. So the new forest growth viewed from the MODIS satellite does not

[20] Searchinger, 2010. Technical note "Comments on draft ILUC analysis of rapeseed and palm, biodiesel presented by E4tECH in February 2010". Available on-line at http://www.endsreport.com/docs/20100428b.pdf

show up as forest, but as one of the "intermediate" categories such as savannah, scrubland or "mixed" (totalling 67% of the land reversion) or even as grassland. That is why the study from E4tech (2010) discussed during the meeting reported only 6% reversion of cropland directly to forest.

- It is clear that satellite data cannot show clearly new forest growth in only 4 to 7 years and the land-use reported by Winrock after just 5 years reversion is not the final land use which should be used to calculate the C stock change. Therefore using this information on intermediate land-cover classes as the final land-cover, systematically underestimates the amount of new forest which is growing and the foregone carbon sequestration from increased EU crop area.

- For a mistake in the datasets published by Winrock International, the coefficients for carbon accumulation (change in biomass C stocks) for reversion of EU land to forest were accidentally set to zero (for comparison, the values assumed for C accumulation on US re-growing forest is 9 t C ha^{-1} yr^{-1}). Therefore, the foregone carbon sequestration was largely underestimated (roughly a factor of 3) for the studies (e.g. E4tech, 2010) which made use of Winrock coefficients to calculate the carbon stored in re-growing EU forests – e.g. on abandoned land.

EPA also used Winrock coefficients, but having realized the above problems, when a natural reversion was occurring in one region, instead of applying Winrock reversion factors they looked at the proportion of land cover in that region (land cover in 2007) and used the annual biomass growth to calculate the reversion of carbon. For example, in a region that was 80% forest, it was assumed that 80% of abandoned agricultural land would grow back as forest.

3.3 GHG emissions from draining peatlands

Carbon emissions from peat oxidation can be measured in two ways: by direct measurements of carbon fluxes on site (mostly CO_2, sometimes CH_4), and by monitoring the subsidence level and determining the relative contribution of oxidation (decomposition and mineralization).

1. GHG flux measurements on peatland are costly and complicated and thus the results are limited. Moreover, it was noted that many studies have not separated root respiration emissions (natural cycling of C) from emissions due to oxidation (caused by drainage) and none of the studies provide the full C balance. CO_2 emissions calculated through flux measurements vary considerably, ranging from below 40 tCO_2 ha^{-1} yr^{-1} [Melling et al, 2005, Muruyama and Bakar, 1996] to well above 70 tCO_2 ha^{-1} yr^{-1} [Hoojier et al., 2006, 2010; Couwenberg et al., 2010]. These GHG flux measurements are best used as an independent check on the results of other methods.

2. The best alternative method to assess peat emissions is by subsidence modelling. Most of the studies show that 75% of subsidence is oxidation, and this can be measured as CO_2 emissions.

The JRC reviewed CO_2 emissions from peatland based on subsidence rate or flux measurements reported in literature (see box below).

Calculation of CO_2 emissions from peatlands

[Couwenberg 2010]:	45 - 90 tCO_2 ha^{-1} yr^{-1} (subsidence, 40% oxidation at 50 & 100 cm depth respectively)
JRC + [Ywih 2010]:	~110 tCO_2 ha^{-1} yr^{-1} (subsidence)
[Fargione 2008]	~ 64 tCO_2 ha^{-1} yr^{-1} (flux)
[Germer & Sauerborn 2008]	~ 33 tCO_2 ha^{-1} yr^{-1} (long-term > 30 yr. **only 2 cm subs. rate**)
Hooijer et al. 2011	86 tCO_2 ha^{-1} yr^{-1} (subsidence; over 200 pts; average first 50 years)
Jauhiainen et al. 2011	92 tCO_2 ha^{-1} yr^{-1} (flux; daytime average over 144 pts; 3 years)
[Wicke 2008]	~39 tCO_2 ha^{-1} yr^{-1}(measurements averaged with IPCC default)
Average	**~ 67 to 74 t CO_2 ha^{-1} yr^{-1}***
IPCC default value	73 tCO_2 ha^{-1} yr^{-1}(2006 guideline)

*** This average includes [Germer and Sauerborn, 2008] value, which refers to a lower subsidence rate of 2cm. Excluding this value, the average is 73 to 80 tCO_2 ha^{-1} yr^{-1}**

The subsidence rate is expected to be 4-6 cm per year in SE Asia, as shown in Figure 10. Germer and Sauerborn, (2008) emissions are based on 2 cm of subsidence rate only). It was shown that the fraction caused by oxidation of tropical peat must be about 60% (Cowenberg et al, 2010), while it is approximately 75% in the Florida Everglades.

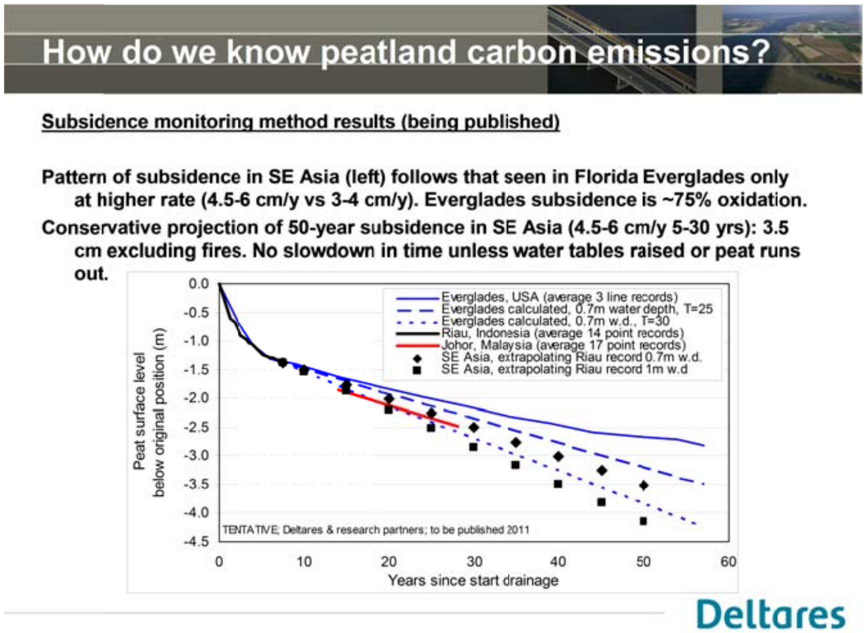

Figure 10: peat subsidence level [presented by A. Hoojier, Deltares]

The IPCC default value for agriculture in the tropical peat falls in the range of averaged values (at the lower limit if we exclude [Germer and Sauerborn, 2008] value, for the reasons explained above). However, this "coincidence" does not mean that the IPCC value for agriculture on peat is the most appropriate to estimate emissions from oil palm plantations on peatlands (for example, it is not unambiguously clear that the agriculture value should be used rather than the managed forest value). Rather, more recent studies [Hooijer et al, 2011, Jauhiainen et al, 2011] were presented at the meeting, based on empirical observations and measurements "in-situ" and providing more robust estimates of the emissions factors.

These studies show relations between water depth / subsidence / carbon loss in Sumatran oil palm and acacia plantations, and suggest carbon loss of **at least 70 tCO_2 ha^{-1} yr^{-1}** at around 70 cm water depth after the first 5 years since drainage (see Figure 11 , combined with Figure 10 above). In the first years this is well over 100 tCO_2 ha^{-1} yr^{-1}, and **annualized over 50 years it is 86 t CO_2 ha^{-1} yr^{-1}**. Concurrent CO_2 flux studies at the same sites find 92 tCO_2 ha^{-1} yr^{-1}. These values are still excluding emissions due to fire, biomass loss and oxidation emissions from non-plantation areas affected by plantation drainage.

Considering all of the above, the emission factor of 20 tC ha^{-1} yr^{-1}proposed by the IPCC in 2006 guidelines (equivalent to 73 tCO_2 ha^{-1} yr^{-1}) should be considered as a still conservative factor for GHG emission estimations, and its application in estimating CO_2 emissions from oil palm plantations on tropical peatland should be discouraged in favour of the more appropriate empirically-based values.

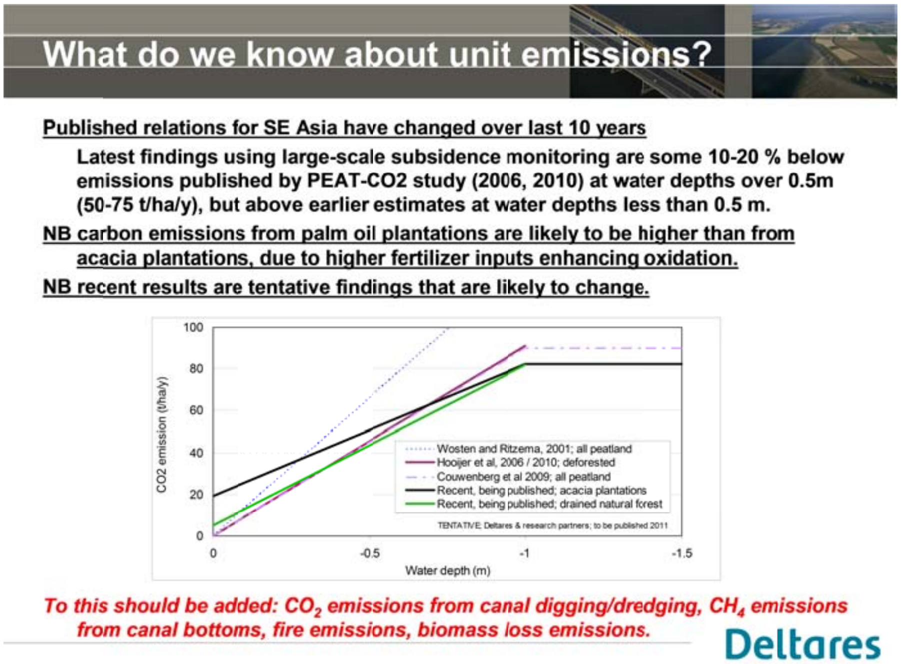

Figure 11: Emissions from peat oxidation as a function of water depth [presented by A. Hoojier, Deltares]

Whilst uncertainties in emissions from tropical peatland have been largely reduced in recent years there is greater uncertainty in identifying the land uses on peat.
CO_2 Emissions from canal digging/dredging and CH_4 emissions from canal bottoms and biomass loss emissions during clearing should also be included. In addition, annual C loss due to peatland fires as a result of drainage must be also considered: annual C loss due to peatland fires in South East Asia is estimated to be about 191 Mt yr^{-1}, and although there are uncertainties on how much can be attributable to plantation development, certainly there is an important correlation, as from Figure 12 below.

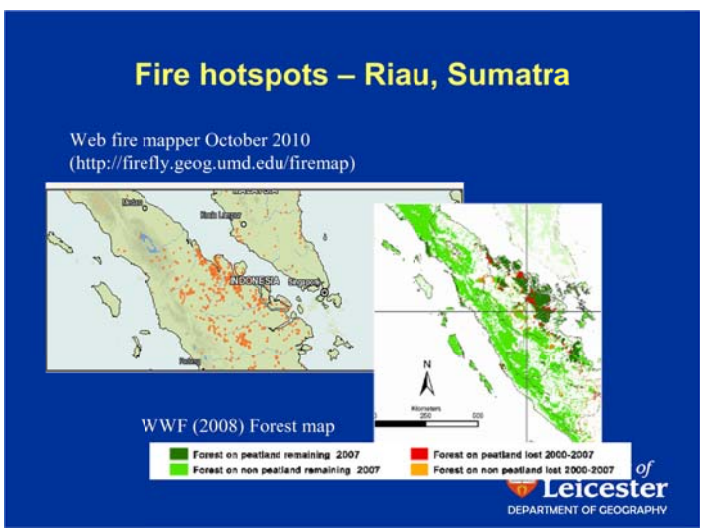

Figure 12: Fire map (2010) and forest map (2007) in Sumatra [presented by S. Page, Univ. of Leicester]

The above emissions (from fires, canal digging etc.), including reduced carbon sequestration, are not accounted for in most of the calculations. An estimate of the most likely values for losses in Carbon sequestration is shown in Figure 13.

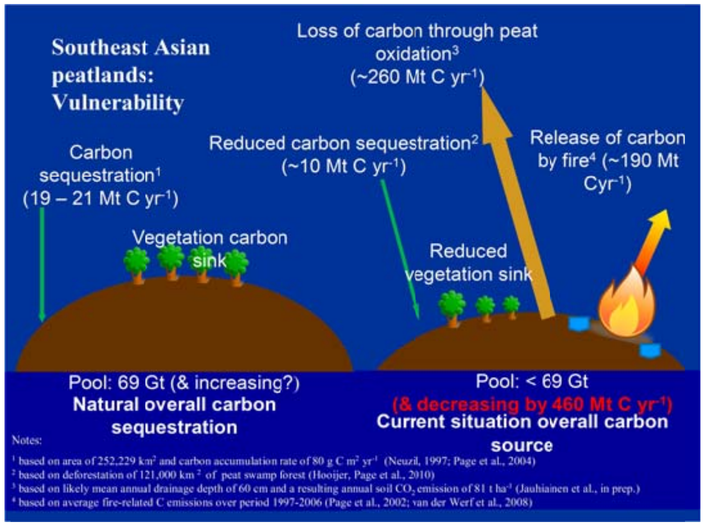

Figure 13: Reduction of C sequestration and C release due to peat oxidation and combustion associated with all forms of land use change on Southeast Asian peatlands [presented by S. Page – Univ. of Leicester]

Tropical Peat Research Laboratory reported that much of the research on peatland is based on temperate peat. Therefore, the different types of peat will influence the decomposition rate and the oxidation levels reported. Tropical peat differs in composition from peat formed by sphagnum and reaction to draining is distinct from the previous.

When special attention is paid to compaction of the peat, the rate of subsidence (and therefore the emissions) may be reduced (but this occurs only on limited areas).The level of peatland drainage is also important: it is the order of 70-100 cm in Indonesia (the target is lower in Malaysia, about 60 cm, but it is unclear what actual water levels are). In any case the IPCC value for tropical peat conversions does not consider deep drainage. It is difficult to assess the areas of peatland, how much C is stored (which depends on the depth of the peat and the bulk density).

The general conclusion is that peatlands are not suitable or sustainable for agricultural production. This is not only because of the carbon emissions, but also because of the inevitability of many of these areas becoming undrainable in the longer term (between 25 and 100 years), and for the loss of biodiversity linked to the switch of tropical peat swamp forest into agricultural area. The representative of TPRL however disagreed with this conclusion, underlying that also social and economic development should be considered.

3.5 Adequate representation of land availability (e.g. CIS and EU)

Land availability in satellite data

It is important to understand if datasets (e.g. FAO, MODIS) are accurately showing land availability since underestimating land availability will affect both the estimation of land conversion and the impacts on C stocks.

It was shown that the low resolution of satellite data (e.g. MODIS 250 m land cover data) makes it difficult to identify changes in land use and thus difficult to estimate how much land is available. Notably, using MODIS data it is not possible to identify important land use changes such as natural forest (e.g. peat swamp forests) being converted to oil palm plantations. To identify these changes in land cover would require much higher resolution data (such as 30 m) which at present is not available globally. However, such data will be available in the near future when the results of an FAO – JRC Global sampling survey are released.

Some tools are available to help the validation of global land cover data, forest and cropland statistics etc., e.g. the Geo-Wiky web tools operated by IIASA[21](an example is shown in Figure 14)

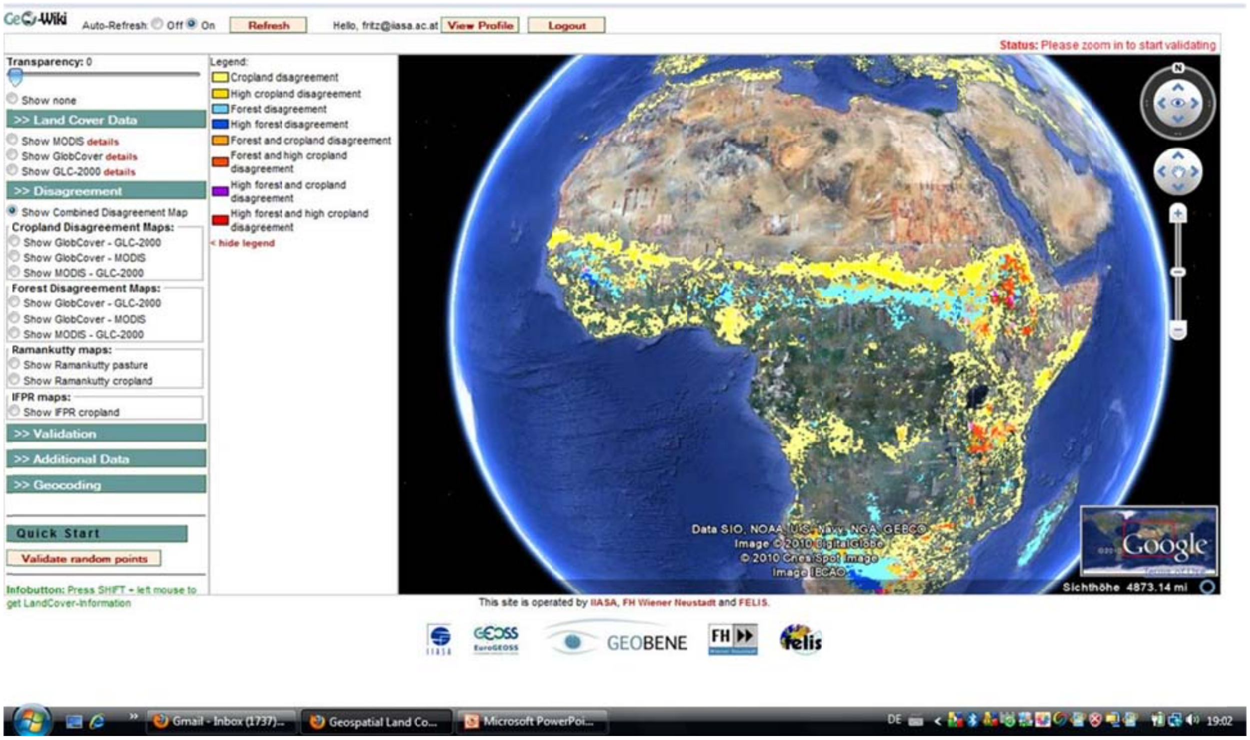

Figure 14: Example of IIASA geo-Wiki tool for Comparison of global land cover products [presented by S. Fritz – IIASA]

"Abandoned land" in CIS and EU

FAO statistics show that since the 1960s cropland has been declining in both the CIS and EU and it was argued that the availability of this abandoned is not always correctly accounted for in the agro-economic models. It was claimed by that the increased demand (for land) due to biofuels will decrease the rate of abandonment of these lands.

Abandoned land may be defined as land that is no longer used as agricultural productive land, and has been converted into other uses for many different reasons. For instance, the abandonment of land in the CIS was caused by the ending of state subsidies, increased labour costs or return of ownership. Formerly drained wetlands in CIS have often been rewetted and returned to forest (to be used in programmes for C sequestration) and would be unlikely to be converted into cropland again. This abandoned land in CIS is not the same as land set-aside in the EU under the Common Agricultural Policy (CAP) which was agricultural land deliberately taken out of production that has now come back into production.

It was noted that that many of the models are not very good at taking into account these structural changes. For example: Partial equilibrium models have regional elasticities that are based on historical price responsiveness of production and these elasticities would not capture the availability of land in Eastern Europe.

Lower yields on abandoned land

In the CIS in particular, yields on abandoned land are lower compared to similar regional climatic conditions. It can be assumed that most of the less productive land was taken out of production (i.e. abandoned) and that the more productive land has been kept in production. Therefore if cropland is increased in those areas, it is highly likely that the increase will be on land which is less productive, with lower yields. Therefore, one cannot assume that extensification in these areas will result in the same yields as on presently cultivated land. It is also important to understand the GHG implications of bringing abandoned land back into back into production. The land use has still changed and therefore there will still be a C stock loss.

[21] available at geo-wiki.org and agriculture.geo-wiki.org

3.6 Uncertainties in datasets

It was agreed that there are great uncertainties in the datasets used to calculate GHG emissions, and these are not always known.

Global land cover maps are of limited use for assessing land cover conversions. The accuracy of the maps being low (each map may have errors that could exceed 25%) the risk is that when comparing two maps, what is measured is the errors in the maps, and not the change in land cover. There are alternative satellite products that do look at change, (e.g. South Dakota State University deforestation maps with a higher resolution of forested areas). More problematic is to find equivalent products (datasets) for agricultural areas.

Winrock's coefficients are normally at country level, and within country, but land allocation is at AEZ level. Moreover in 2020 there might be no forest left in some regions and therefore country-based historic data may no longer be realistic.

In any case reliable historical data are considered important to verify the land conversion rates, in particular in hot spots; good quality data at regional scale are available (at least for some regions as SE Asia) and need to be used rather than poor quality global data.

In conclusion it was agreed that while there are uncertainties in the datasets, all indicators point towards the existence of a significant ILUC effect.

4 Agro-economic modelling and uncertainties:

This section focuses on the lack of empirical information available, where exactly the models need more information and which important relationships are not being modelled. There are many decisions that have gone into the existing agro-economic models that can cause uncertainties in the projections. For example a single change in the underlying baseline conditions in a simulation could change the amount of biofuels being used, where they are being used and where the biofuels are being sourced from. There are uncertainties coming from data on land use change, yields and fertilizers for example, which are all very important, but underlying these factors are uncertainties inherent in the whole modelling exercise. Despite these uncertainties, it is recognised that because economics are driving the whole 'ILUC' process the use of agro-economic models is essential to the estimation of ILUC.

Economic market models are constructed in order to describe and simulate national and world markets for commodities, manufactured goods and services. When they focus on the production, market and trade flows of agricultural commodities *only*, ignoring the rest of the economy, they are classified as *partial equilibrium* (PE) *models* and known as 'agro-economic models'. When economic market models depict *all* commodity, goods and service sectors (including agriculture), together with the interactions between them, these models are known as *general equilibrium* (GE) *models* and the name 'economic market model' is appropriate.

Models of both kinds are referred to in this report: for example, AGLINK-COSIMO, CAPRI and ESIM are PE agro-economic models whereas GTAP, LEITAP and IFPRI-MIRAGE are GE models. Although each type of model has strengths and weaknesses when it comes to simulating the global effects of a policy involving a particular group of agricultural commodities, they all share two important characteristics. First, their prime focus is on markets (behaviour and outcomes) and hence they are usually specified at the level of aggregation that defines the market (i.e. national[22]). Second, the underlying driving forces are those of rational economic market behaviour (e.g. higher prices encourage supply but discourage demand, markets reach equilibrium, supply is encouraged where production cost is low and vice versa) but they do not shed light directly on the micro-decision making of particular kinds of producers or suppliers, or on resource use generally. Thus, prices – not agronomic or technical possibilities - play a key role in determining how and where production occurs. At the same time, the behavioural coefficients and elasticities in these models reflect a *large, but hidden*, information set consistent with the relevant underlying physical production conditions, and additional variables (e.g. for yield growth, technological change) are usually included to allow these conditions to vary over time. A first source of error is, therefore, the way economic behaviour is depicted in the model: the causal economic relationships depicted and the parameters used. However, when used for their more normal tasks, users (policymakers, analysts) consider that this level of uncertainty is acceptable.

To extrapolate from production decisions to land use, each model employs its own method, usually based in some way on average yield coefficients (which allow the calculation of how much land of 'average' quality would have been needed to produce a given quantity). This raises the question of whether marginal yield, for a specific crop in a specific location, should perhaps be lower or higher than the average yield. Clearly, the answer depends largely on which type of land is used for crop expansion (previously cropped land – of higher or lower quality? permanent grassland, degraded land, virgin rain forest, peat land etc.[23]). The data required to model this in a rigorous way are not available to the typical economic market model, and certainly not with global coverage. Therefore, the quantitative estimates of land use change produced by these models are approximate and subject to a high degree of uncertainty. This uncertainty in the land use change estimates is transmitted to any estimates of GHG emissions based upon them.

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[22]Attempts have been made to regionalize some of these models: for example, CAPRI can be run with the EU disaggregated to NUTS 2 level (over 220 EU regions), GTAP can be linked to AEZ, a global land use model that distinguishes 18 different agro-ecological zones.

[23]Typically, economic market models do not model *total land* (including all its uses, commercial and non-commercial). In general (for example, in GTAP), the total land area used for crops, pasture and commercial forestry is forced to remain constant. This means that price-induced increases in cropland must be at the expense of pasture or commercial forests, and the depletion of rainforests or other ecologically-valuable non-commercial land cannot be simulated. In AGLINK-COSIMO, 'total land' is land used for agriculture, and does not include commercial forestry.

Summarising the above, the ILUC results of these models contain various uncertainties that derive from their specification and the fact that when asked to simulate ILUC they are being pushed to the frontier - or beyond – of the range of tasks they were designed to perform. All models contain an element of error (uncertainty) due to the fact that they are stylized representations of reality. However, when used for their normal tasks, this level of uncertainty is usually considered acceptable. A second layer of uncertainty, potentially far greater, is introduced when the core model results are used to extrapolate to ILUC, using less rigorous methodology than that used for the construction of the model itself, with inadequate empirical content and often, it should be added, with inadequate behavioural knowledge of the reactions of micro-agents in the real world. A third source of uncertainty has not yet been mentioned, as it arises from the way these models are *used* rather than from their underlying specification. In general, in order to estimate the future effects of a policy, an economic model is simulated over a given future time period without the policy change, and again with the policy change, holding everything else constant. The effects of the policy change over the period, or in the closing year of the period, are measured as the differences between the two scenarios. This means that each model exercise needs to use assumptions about the evolution of exogenous (external) factors over the time period (e.g. GDP, population, exchange rates, technological advance, productivity (including agricultural yields) and so on[24]). These choices are a source of further uncertainty, and can cause differences between the results of different models. For example, a simulation study based on a vision of rapid recovery from the global economic crisis (and hence higher GDP, higher transport fuel use, higher fossil-fuel prices etc.) may give noticeably higher ILUC impacts than when more a more pessimistic outlook is adopted. Similarly, a view of future technological and yield developments based on those observed up to the mid-1990s will give different results from those incorporating a more pessimistic vision incorporating climate change-induced increases in desertification and water shortages, and sluggish yield growth. This type of uncertainty cannot be substantially remedied by more data or greater understanding of the behaviour of particular agents: it arises because the future cannot be fully predicted.

The main strength of global (agro-) economic models for generating estimates of ILUC is that the resulting estimates are fully compatible with a consistent and detailed global view of production, consumption, trade and prices for the relevant commodities, and with the economic behaviour that drives these outcomes. This provides an invaluable background against which to check the plausibility of the ILUC estimates, which would be lacking in models based solely on agronomic principles. On the other hand, this strength cannot offset the considerable uncertainties in the ILUC estimates generated by these models.

4.1 Intensification

4.1.1Impact of pasture converted to cropland

Many agro-economic models, with the exception of GTAP based models, assume that if pasture is lost to cropland there is no expansion of pasture onto forest as a result. Thus the models estimate the minimum amount of ILUC from this point of view. GTAP-based models include knock-on effects on pasture area indirectly by modelling economic competition for land between crops, pasture and commercial forestry.

1. FAPRI models the competition between cropland and pasture with substitution between the two.
2. In the LEITAP model there is substitution between the feed and the grassland used, but there is large uncertainty about the substitution elasticities.
3. In the IFPRI-MIRAGE model pasture is replaced by feedstock: When there is no more complimentarity between feedstock and land you start to have substitution, which leads to potential intensification or extensification depending on the price response of the land vs. feedstock. With an increased demand for agricultural crops the price of crops increase, the

[24]In GE models, some of these assumptions are implicitly set by the closure rules – but choice of closure rules is, also, open to discussion. Typically, PE models need a larger number of quantitative exogenous projections in order to run.

price of intermediate inputs rise, production costs go up, demand of meat goes down and therefore less land is used. If the aggregate price of the feedstock goes up, extensification will result, but if the aggregate price of the feedstock goes down, intensification will occur. This aggregate price depends not only on the price of wheat or corn but also on the by-products. IFPRI-MIRAGE shows intensification in the US and EU, but extensification in Brazil.

4. The BLUM model for Brazil presented by ICONE (and used by FAPRI) calculates the displacement of cattle by agricultural crops. The model includes pasture, cattle structure, and beef production, which are the main variables needed to determine the performance of the cattle sector. It was noted that the slaughter rate must also be included in the models, because if this factor is not included then the demand for pasture land could be under or overestimated.

It was stated that whilst the cost of animal feed may go up as crop prices rise, some of the biofuel by-products are used for animal feed, so the price of feed will come down. The result is that less soybean will be grown, as this is primarily grown for the meal, and this will affect the total animal price. Therefore, to calculate the net livestock count (herd size) all the elasticities have to be accurate.

Historical data and model assumptions

To calibrate the elasticities in the models good historical data on pasture area are needed. However, many of the pasture effects are highly regional and validation of the elasticities and modelling results is often difficult, even for countries like Brazil, where regional data on changes in cropland, pasture and cattle numbers are readily available. In fact, agro-economic models simulate market behaviour and are usually specified at the level of the market (i.e. national). Some of these models then break down the results to the regional level, because many of the effects of interest are very region-specific, even local, and this step is often a big challenge for market-based agro-economic models

The historical data used to calibrate elasticities in the models is not always representative of the present and future trend. For example one of the key assumptions in the BLUM model is the herd size. ICONE showed that in the US beef production is increasing whilst herd sizes are reducing, but in contrast beef production in Brazil has been increasing along with increasing herd size. However, since 2005 the herd size in Brazil has been stable and in the future, cattle herd sizes are likely to reduce and expansion onto pasture may not be required. Thus, even with the best historical data available in Brazil, ICONE still had to make several assumptions in the BLUM model.

Given the uncertainties in measured data a cautionary approach should be taken when using measured data to derive elasticities. It can be said that modelled results are really being driven by the assumptions in the model.

Deforestation

The BLUM model accounts for deforestation. ICONE showed that even though total cropland is increasing in Brazil, deforestation does not take place in all six regions of Brazil. In some regions the expansion may be on savannah or other land types. Due to the lack of data it is not known if the model is underestimating or overestimating deforestation. Therefore, the land supply elasticities need careful validation. The model applies different crop price elasticities for the main land uses that compete for land (sugar cane/soy bean) within six regions in Brazil. The land supply elasticity is low for regions where the land availability is low and high where there is high land availability (i.e. the frontier) and the data are calibrated by satellite imagery.

In the BLUM model, the expansion of cropland in the Amazon is mainly a function of the beef price, whereas the deforestation of the savannahs is mainly a function of the soybean price. What is not known is how much these prices will increase. It was suggested that if biofuel co-products reduce the price of feed and beef then the rate of deforestation would decrease. However, most of the available models show that this decline in price is not happening.

4.1.2 Contribution of double cropping to increased production

The first response to a crop price increase can be to double crop. Intensification through double cropping can be considered a low cost source of increasing production. In an example shown by Iowa State University, double cropping in the US has increased in recent years (See Figure 15), as the profitability of double cropping increased with higher crop prices. To put this in perspective, in 2008 the area of soybean acres that were double cropped in the US represented approximately 9% of the total US soybean area (FAO, 2011)[25].

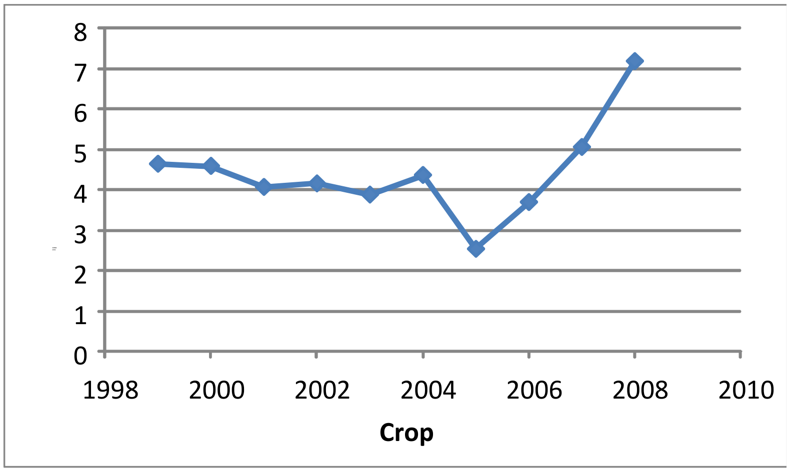


Figure 15: Number of double cropped soybean acres in the US (million acres) [presented by J. Fabiosa –FAPRI CARD]

It was discussed whether double cropping can be considered sustainable and whether it really indicates an increase in crop production.

The main double cropping system in the US is soybean after winter wheat and this system is considered sustainable in the US regions where double cropping is already taking place. However, in response it was noted that in the US, double cropping is limited to a few states, where this system is already typical cropping behaviour.

The difference in yields between a long soybean growing system and a short double cropped soybean system may also have an impact on whether double cropping increases total crop production or not.

4.2 Yields

4.2.1 Importance of yields in models

To see whether the implementation of a biofuel scenario has a positive or negative yield effect it is important to understand how an economic model behaves with regards to:

- What is weighted to marginal productivity of land?
- What is the composition effect with regards to crop shifting and land allocation?
- What is the marginal productivity on land for cereals within one region (e.g. one AEZ in the US)?

There is a great dependence on where the land expansion is going to take place (e.g. in a region such as

[25]FAOSTAT | FAO Statistics Division 2011 | 04 April 2011 shows total US soybean area in 2008 was 30,222, 000 ha = 74.8 million acres.

Eastern Europe where the yield is initially below the average). In this case the biofuel scenario will reduce the average yield. In other cases the expansion may take place in a more competitive region where a lot will depend on the land suitability or availability within the region.
Crop shifting (e.g. wheat to corn) would show a large increase in yield of corn in the model. However, it is important to note that this yield increase has nothing to do with fertiliser, or marginal productivity of land, and is just a shift of land use. It is more essential to look at how the yield of just one crop (e.g. wheat) reacts in the model. In addition, it is also important to understand how the models calculate the yield growth on existing cropland, and not just the yield slippage factor.

4.2.2 Meeting biofuel demand through yield increases

Meeting the demand for biofuels without expanding cropland would need significant increases in yields. In Figure 16 the green bars show the worldwide trend in the rate of yield increase (e.g. 1.2% for cereals). The blue bars indicate the rate of yield growth you would need to meet global demand without expanding cropland area. The red bars shows the additional yield increase you would need to provide all the food demand and the biofuel demand without expanding cropland (e.g. total of 2.4% for cereals).

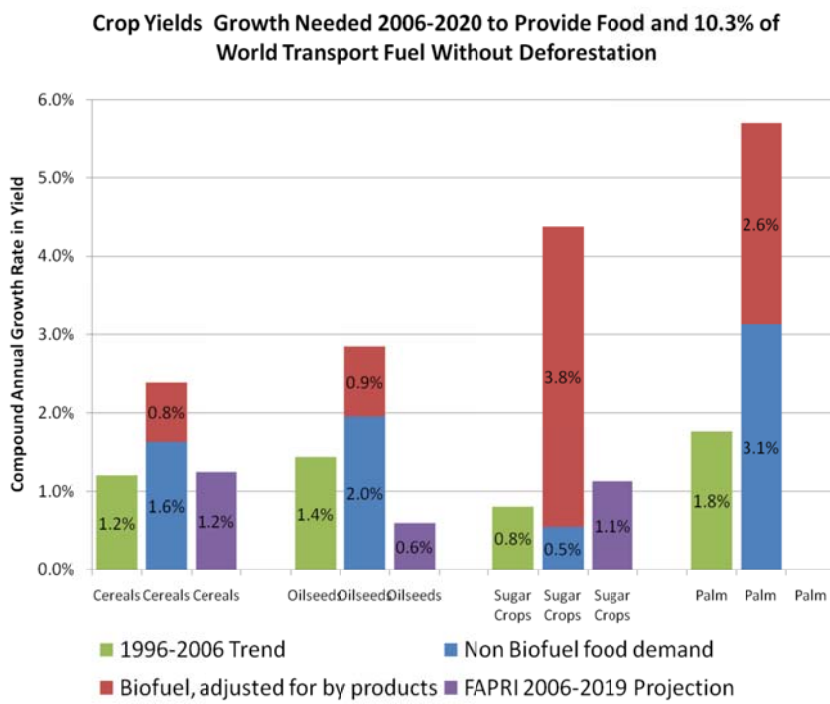


Figure 16: Crop yields growth needed to provide food and biofuels without deforestation [presented by T. Searchinger – Princeton University]

In order to meet both biofuel and food demand increases without land expansion between now and 2020 it would be necessary to double the rate of yield increase for cereals. Figure 16 also shows that oilseeds, sugar crops and palm oil would need even greater increases in yields.

Increasing fertilizer rates to increase yields

Models such as IFPRI-MIRAGE have a logistic function that relates yield to the level of inputs (all fertiliser use, capital labour etc.) but calibrating these functions is difficult. Looking solely at the relationship between the application of N and yield does not give the full picture as there are many alternative ways to increase yields including the use of new crop varieties, fungicide treated seeds, pesticides, increased mechanization, improved timing of inputs or land improvements.

There is a valid correlation between changes in crop price and fertilizer application for US corn, but the correlation between crop price and yield is uncertain. In Argentina the application rates of Nitrogen (N) fertiliser use were relatively low compared to Europe. However, in the past four years there has been a trend of increased N fertiliser use in Argentina which correlates well with the increase in prices (according to IFPRI). Although the application rates are still far below European usage, they are increasing faster than many other places. However, it was added that this increase may also be the consequence of other agro-economic decisions.

Because application rates can be adjusted in the short term farmers would tend to increase fertilizer use in the belief that they are achieving higher yields. It has been argued that there is an opportunity to increase fertiliser rates and thus production in places where current fertiliser application rates are low (note that this is a one-off catching up process. Moreover, it has to be economically profitable. FAPRI showed that 20 % of the total cropland area[26] in the world used for corn in 2006 to 2007 had N application rates of less than 50 Kg N/ha (see Figure 17).

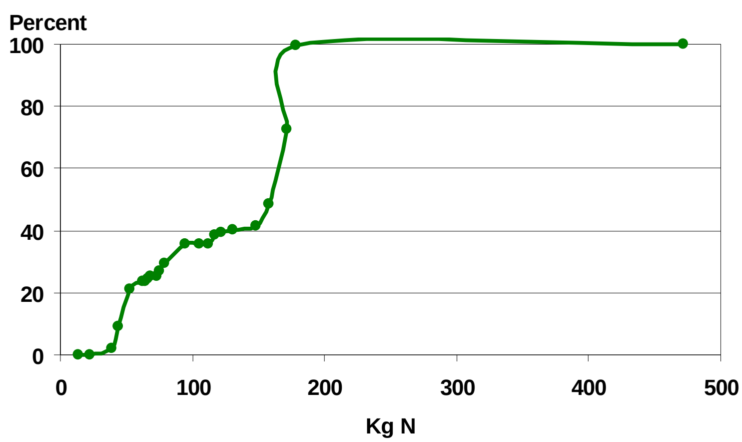


Figure 17: Corn area and (N) fertilizer use [presented by J. Fabiosa – FAPRI CARD]

However, the true opportunity to increase production may be constrained more so by water availability and/or soil quality and the extra yield gain which would actually come from changes in these factors could be incorrectly attributed to the increase in N fertiliser. It is also important to consider the possible C changes that could arise with the intensification of N fertiliser use.

In many developing countries yield increase can be related to improved N efficiency, with reduced emissions per tonne. It is not exactly clear what a price signal would do to N fertiliser application rates in Africa, because application rates in Africa are closely related to other factors such as availability, market structure and distribution problems.

[26] Total area in the study is only 75% of the world corn area (FAPRI).

4.2.3 GHG emissions from increased fertilizer application

Where the application of N fertilizer is primarily used to increase yields, this will increase the GHG emissions per tonne of production because of diminishing returns, particularly where the additional N is applied on already intensively fertilized crop areas[27]. In some cases these emissions could exceed the LUC emissions. Also the FASOM model used by EPA for domestic agricultural modelling reports the emissions caused by more intensive management with respect to extra N application.

Economic theory shows that the marginal spending (Euro/tonne) on farming inputs is much higher than average spending on inputs. So if the distribution of spending between inputs stays the same one would expect higher emissions per tonne.

Marginal emissions per tonne from inputs could only be lower if there was an enormous swing of marginal spending away from high-emission-per euro inputs like fertilizer towards low-emission inputs. This would mean in practice that farmers would use no more fertilizer if crop prices increased. However, there is a clear link between fertilizer spending and crop price.

4.2.4 Comparison of yields on new cropland to yields on existing cropland

Yields on new cropland

Yield obtained from a given area of land is the product of two factors:

(1) The yield gap (i.e. percentage underperformance of the actual output relative to the theoretical or demonstrated maximum)

(2) The crop-specific yield potential of the land linked to certain aspects of the soil composition, local climate and other site-specific factors.

In the EU, much of the land that is potentially available for expansion is land that was previously abandoned and historically yields on this land were low. Even in 2008 the prices were not high enough to make this land profitable to use. JRC-IE presented FAO data that shows the farms where land was abandoned had only approximately 28% of the EU average wheat yield. This does not even include the ratio between worst and average yield on fields on one farm, which for England is reported to be about 0.65, making marginal cereals yield possibly as low as 0.18 of the EU wheat yield if this figure also applies to the marginal EU farms.

However, it was argued that yield potential and yields gaps for each unit of land are not fixed and that in the period since the land was abandoned there may have been developments that could increase the potential yields. Furthermore yield investment in inputs, machinery, land improvement and agronomic expertise may reduce the yield gap when the economic situation supports these investments.

It was demonstrated by ICONE that during the first growing seasons on new land the yields of sugarcane in Brazil may well be below average, but after a few years, when the technology has been adapted to the local conditions, average yields can be reached. However, as described later in this section in certain conditions the yields of sugarcane on new cropland may also be higher than average but can decrease once the land is in use.

FAPRI showed at the workshop that in specific counties within the US there is no evidence of large yield changes due to cropland expansion (See Figure 18). However, this does not take into account what happens within the county

[27] ENSUS indicated that increased yields in UK between 1997 and 2008 were obtained without increased use of fertiliser per tonne.

Commodity	No Expansion Yield	Yield in Expansion Counties	Ratio
Wheat (bu)	40.5	49.8	1.23
Potatoes (cwt)	426.9	519.8	1.22
Peanuts (lbs)	3244.8	3622.6	1.12
Barley (bu)	60.3	63.4	1.05
Canola (lbs)	1537.3	1567.3	1.02
Rice (pounds)	7141.3	7014.0	0.98
Cotton (lbs)	914.3	886.4	0.97
Corn (bu)	158.7	151.4	0.95
Rye (bu)	19.3	18.0	0.93
Beans (lbs)	1726.7	1584.4	0.92
Sugarbeets (tons)	26.8	24.0	0.90
Sorghum(bu)	70.8	60.8	0.86
Oats (bu)	62.3	52.6	0.84
Soybeans (bu)	43.5	35.7	0.82

Figure 18: **Yields in US specific counties with or without cropland expansion [presented by B. Babcock – FAPRI)**

EPA indicated that in the Conservation Reserve Programme (CRP) in the US, farmers set aside just 10% of the cropland and as a result total production went down by much less than 10%.It was implied that this indicates that the marginal land has lower productivity than the land kept in practice. It was argued that the CRP programme was specifically designed to encourage farmers to take poor land, at risk or highly erodible, out of production. Thus, farmers enrolled the least productive fields with the lowest yields.

Models assume a ratio of marginal and average productivities that measures the productivity of new cropland versus the productivity of existing cropland(Tyner et al. 2010).Apart from GTAP-based models (excluding LEITAP), all the models take the yield on new cropland to be within a few per cent of the average yield. In contrast, the GTAP and IPFRI-MIRAGE models make the assumption that the marginal productivity on new croplands is significantly lower than average productivity.

Previous versions of GTAP-Purdue (Tyner et al 2009) assume a ratio of 0.66 for cropland all across the world, independent of the previous use of the land, whilst IFPRI-MIRAGE takes an overall factor of 0.5 and specifically for Brazil a factor of 0.75[28] (IFPRI, 2010).

In contrast, the Brazilian Land Use Model (BLUM) model developed by FAPRI and ICONE suggests that in some cases initial productivity on new cropland can actually be greater than existing cropland. This can be justified for some cases in Brazil, where the land already exploited is limited by access rather than suitability, and where old land is sometimes degraded. In Brazil the land available for new cropland (previously deforested land) often has high initial productivity but due to poor agronomic management there is often degradation and in time the reduction in productivity can force farmers to find new cropland with a higher yield.

ICONE compared 2001 and 2007 Brazilian yield data for several crops in counties where agricultural production had occurred and in counties with established cropped areas. The results showed factors of 0.95 for sugar cane, 0.97 for soybean, 0.91 for corn, approximately 1 for rice, and slightly more than 1 for pasture. ICONE estimated that Brazil has 30 M ha of native vegetation land with high and medium suitability for annual crops, which should give productivity close to the average. Most of this suitable land is on savannahs, whilst the Amazon was excluded. ICONE also reported 60Mha of pasture could be converted into cropland, with productivity also close to the average. However, ICONE also reported that with time the productivity on the new land will decrease, to 0.5 or 0.7 which is closer to the values reported by the GTAP or IFPRI models.

[28] The IFPRI factors apply to new cropland coming from pristine ecosystems, primary forest and savannah, but not to pasture.

Global estimations of land availability and productivity

Global estimates suggest that new land could have theoretically higher yields than on existing land. Tyner et al. (2010) used a set of regional yield ratios at the AEZ level, obtained from a bio-process-based biogeochemistry model (Terrestrial Ecosystem Model (TEM): Zhuang et al., 2003) along with spatially referenced information on climate, elevation, soils, and vegetation land use data. The results show that the yield ratio varies across the world and among AEZs.

However, it was argued that TEM Method systematically underestimates the difference between average and marginal yield. The problem is that the TEM method assumes that the only source of yield variation is land suitability, which is uniform over areas of about 2500km (which implies hence that also yield is uniform over these areas). However, the drivers of suitability, such as soil type or climate, in agricultural land may be inhomogeneous within a 2500km grid cell (land suitability varies typically on much smaller scale outside the US). By ignoring all other sources of yield variation (like variation on individual farms, variation due to levels of competence and investment, variation between farms in each 2500 Km2 grid), TEM might underestimate the difference in average and marginal yield.

In regions such as South America there is a substantial amount of land that could be above the previous GTAP ratio of 0.66.The full ratio from average political-regional yield to marginal field yield needs to account also for the variation in the yield between AEZs in the region and the ratio of yields inside the grid-square. Globally within each agricultural zone there is a large quantity of land that has a higher productivity than the land actually being used, and in many zones even the average productivity of the land that is not being used is higher than the productivity of the land being used. However, reported yields (e.g. in the SAGE M3 database) indicate that areas in the world with the higher yields correspond well to the areas with the highest agricultural use, which confirms that the best areas/soils are already in use. Moreover, a global biophysical analysis such as TEM does not account for additional reasons why farmers cannot access land that may be more suitable and more productive.

There are many other essential factors that the global biophysical models cannot take into account, such as regional soil properties. In Brazil savannah soils are not considered suitable for agricultural, due to low fertility. However, in reality the savannah has been converted and the soils corrected for agricultural use (through the implementation of new technologies).

4.2.5 Crop displacement

Regarding changes in use on existing cropped areas (crop displacement to meet demand) models assume that if a low-yielding crop (e.g. barley) is replaced by a high-yielding crop such as wheat, the yield will jump from the regional average barley yield to the regional average wheat yield.
The key findings of a FAPRI-CARD study (shown at the workshop) indicate that in the US, the shifting of crops (displacing one crop for another) has been much more significant than expansion of crop land (See Figure 19). A 50% increase in expected farmer returns from growing principal crops led to a 1.7% increase in acreage from 2006 to 2009 (about 4 million acres) indicates that U.S crop acreage is inelastic.
It was argued that this study gives the impression that the increase of demand leads to crop displacement which benefit the yields.

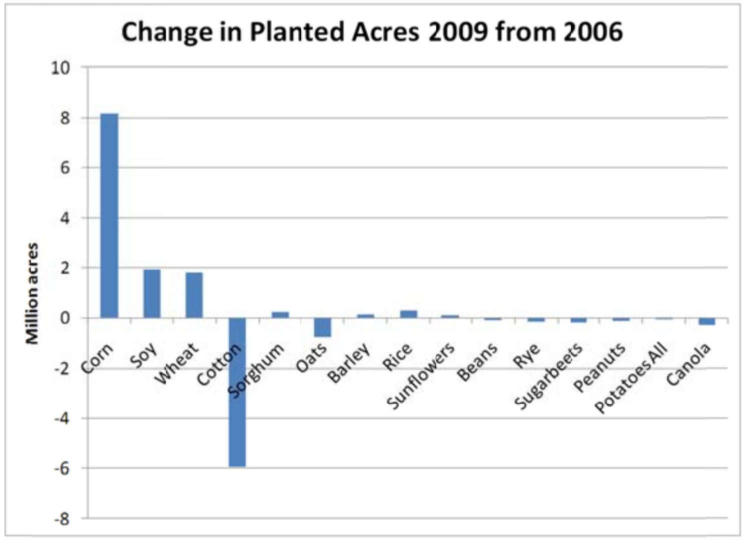


Figure 19: Change in planted acres from 2006 to 2009 in the U.S. [presented by B. Babcock – FAPRI)

There was disagreement whether crop shifting results in significant changes to yields or not. It was stated that farmers tend to grow which ever crop is going to give them the best yield on that particular land. Therefore, it is wrong to assume that if a farmer switches from barley to wheat the yield of that wheat crop will be the same as the average wheat yield. However, it was also stated that typically farmers have used a rotation of wheat/barley/rape to prevent disease. With improved varieties of wheat farmers have moved to a wheat/wheat/rape rotation because they can get a higher yield on the second wheat crop than barley. When farmers see high prices they are more inclined to change the crop rotation.

4.2.6 Effect of price on yields and rate of yield increase

Crop-price effect on yield (the yield elasticity on price)

This is a short-term (1-5 years) reversible effect, due to changes in farm inputs, and all models include it. However they do not consider the extra emissions from extra inputs. The world's farmers are not aware of biofuels mandates; farmers experience price changes. Therefore without an effect on food price, there can be no increase of yields due to biofuels mandates.

The JRC –IPTS biofuel study (JRC, 2010a) used three partial equilibrium models: AGLINK-COSIMO, ESIM and CAPRI. In all three of these models yield is responsive to price. The underlying assumption in the models is that in the yield functions, yield can respond with quite small coefficients to current price and price with a one year time lag. The assumption is that if prices were or are much higher farmers can respond in the short run (e.g. increase fertiliser or increase feeding rates). However, if prices go down again then yields in subsequent years will fall again as these responses are not cumulative.

Non-reversible crop-price effect on rate of yield improvement (research spending and long-term farm investment effect).

The most optimistic assumption is that the rate of yield increase is proportional to crop price (no diminishing returns), and there is no time-lag before effects are seen. In theory if prices go up then R&D should go up too, but historically prices are very persistent over time. Prices have gone up and down, but there has been little yield response to price changes. The FAPRI-CARD model has a lower effect whilst other models do not include it.

There is approximately a 20 year time-lag before research results on plant-breeding for yield increases are seen in yields, but on-farm investments will often have less time-lag. Beyond 2020, 1st generation must be a long term commitment, and not just a step to 2nd generation. It is therefore important to know the life time of

1st generation biofuels (e.g. 20 years or 30 years) and what will be the mandate. There is a reasonable chance that there will be a meaningful effect on yields beyond 2020, but it too uncertain to be taken into account.
Two thirds of agricultural research spending is by governments, but this proportion may decrease with increasing price (more public support for farmers is needed when profitability is lower). It is also important to differentiate between public and private R&D as the adoption rate of technology is very different.
In recent years agricultural improvements have improved the efficiency of fertilizer use in developed countries, which is seen against a background of real crop prices falling and increasing fertilizer prices. Increasing crop prices with biofuels policy would redirect research to increasing yields rather than reducing costs and the effect of such an increase in prices on farm emissions is not clear.
Cumulative changes in yield come from the trend term in the yield functions, which increase gradually over time. JRC-IPTS study compared scenarios with and without EU biofuel policy, (leaving the biofuel policies of other countries unchanged). The impact of EU biofuel policy on feedstock prices on the world market was very low (3-4%). The question is whether R&D will react to such a small increase, and increase investment into yields or whether the stimulus will come from increased food demand.
There are other reasons for R&D apart from biofuels which are climate needs, adaptation needs and mitigation. Economic models are calibrated to historical data during which weather certainly varied, but *climate* was relatively stable. If the extreme weather experienced globally in 2010 is an indication of what's coming in the next decade and beyond, existing models will tend to under-estimate yield in some areas and over-estimate yields in far more locations. For example, based on a probabilistic analysis of climate change and its effects on crop yields, Tebaldi and Lobell (2008) write: "We estimate the chance that global losses from climate change by 2030 will outweigh gains from CO_2 as unlikely for wheat (<33% chance), likely for barley (>66% chance) and virtually certain for maize (>99% chance). In addition, we estimate larger than 80% chance that net losses for maize will exceed 10% over this relatively short time period." Lobell and Burke (2010) note that "though most cropping systems exhibit a clearly negative yield response to warming, the precise amount of yield loss per degree warming is often not tightly constrained, either from theory or observations."
Therefore, although it is not known how effective research that targets these needs will be or how much it will actually address the issues raised by biofuels, the longer the analytical horizon used to estimate the climate effects of fuels, the more important it becomes to consider the potential effects of climate on agriculture.

4.2.7 Yield increase vs. Area increase

Increased supply for crops is met by a combination of increased land area and increased yields. Figure 20 shows that in developing countries most output increase since 1961 has come from area increase, but this trend is reversed in developed counties.

Region	Cereals EU	Rapeseed EU	Maize US	Soy S Am	Sugar cane Brazil	Oil Palm S E Asia
Yield growth change / output growth change	78%	37%	58%	10%	23%	23%
Area growth change / output growth change	22%	63%	42%	90%	77%	77%

Figure 20 Rate-of-area increase against rate-of-yield increase (Presented by W.Lywood – source FAOSTAT).

In response it was agreed that in Europe most of the extra supply comes from yield increase because the CAP has prevented the area from increasing. The slippage factor (as discussed in section 4.2.4) must be taken into account when considering the extra cropland in the biofuels case.
It is debatable to what extent this conclusion would hold for the effect of price increases rather than time, but the general trend seems reasonable: although there is more possibility to increase yields in developing countries (higher yield elasticity) there is also much more extra land available (much higher area elasticity).

It was commented that the focus in many studies has been on what the yield growth has been, and then attributing the remainder to land area growth. It was proposed that it would be more appropriate to model

the land area growth, and then allocate the rest from yield growth, since that accounts for nearly all the increase in supply growth. Analysis of historic data shows that: area growth and yield growth are both driven by price. There is a danger in estimating area and yield elasticity separately without considering their ratio (which is the area change/yield change). Recent GTAP modifications have taken into account the first effect but not the second, coming to the conclusion that in developing countries *less* of the extra production will come from increased area than in developed countries.
Almost all the models included in the JRC comparison study reported substantial yield responses, for example:

- GTAP maize ethanol – 43%
- IFPRI-IMPACT corn ethanol - 80%
- IFPRI-IMPACT wheat ethanol – 70%
- FAPRI CARD EU wheat ethanol – 15%
- GTAP Rapeseed biodiesel – 40%

These account for the yield gain through price but also the yield loss through expansion onto marginal land.

4.2.8 Uncertainty in yield elasticities and risk for policy

Correct analysis of statistics (empirical data) indicates that yields are much less elastic than assumed by models like GTAP. The problem is that prices depend on both demand and supply. To separate the effect of price on the subsequent (lagged) supply, it is essential to look at the effect of a variable (instrumental) which only changes supply or demand.

Price is not a suitable choice, but weather shocks are. A rigorous statistical analysis of the effect of weather shocks on total crop supply (presented by Wolfram Schlenker at the workshop) shows that the elasticity of supply is roughly twice the elasticity of demand. The supply elasticity of 0.12 is mainly due to area elasticity whilst yields are rather inelastic (0.04). That means that increase in crop demand from biofuels will mostly be met by increases in crop area.

Although yields have been increasing over time, what cannot be said with certainty is:
(a) What process has been driving that increase?
(b) Whether it will continue in the future?
(c) At what rate?

Therefore there is a risk in basing biofuel policy on assumptions about future yield growth occurring at a particular rate.
An important outcome of the workshop was that, in a case like this, sensitivity analysis is required to see just how sensitive policy outcomes ARE to different rates of growth, and to establish the worst-case possibility.

4.3ILUC reductions

4.3.1 ILUC reduction from by-products

The assumptions made by agro-economic models regarding by-products can have a significant effect on the amount of ILUC reported by the models. For instance, the protein content of biofuel co-products can reduce the demand for meal and thus affect imports of soybean that is the main marginal global source of high protein animal feed.

It is essential that the models differentiate between fodder energy and protein feed as farmers make a balanced feed plan based on the appropriate amount given to the animals. The marginal source of fodder energy (mainly grain crops) is very different from the marginal source of protein (soy meal). They have different fertiliser application rates and yields and will have a different impact with regards to indirect land use change and environmental impacts.
The ratio of protein vs. energy varies significantly in different oil meals: soy meal has high protein content, rapeseed medium content and palm kernel oil meal a very low content.
The key differences between Soy meal and palm oil are presented below:

Soy meal is grown primarily for its meal:

- Meal accounts for about 60% of the value of soy bean
- The oil yield of soy is only about 10% of that of oil palm
- Soy meal accounted for 83% of trade in high protein meals from 2006-2009
- Soy meal accounted for 91% of growth in trade in high protein meals from 1999 to 2009

Palm oil is grown nearly entirely for its oil and is the marginal global source of vegetable oil

- Oil accounts for about 98% of the value from oil palm
- Palm accounted for 80% of growth in trade in high protein oils from 1999 to 2009

It was demonstrated that an LCA approach can capture these significant differences between fodder and protein sources[29]. For example, if more rapeseed oil is required to meet biodiesel demand, then more rapeseed meal will be produced, and thus more protein and more food energy will be available. The protein from the rapeseed meal displaces some soy meal (co-produced with soybean oil) so less oil will be produced. This reduction in oil is compensated by marginal vegetable oils (e.g. palm oil). The LCA calculation for solving this considers the amount of rapeseed oil required, and then accounts for the co-products, the marginal source of oil (palm oil), the marginal source of protein (soy meal) and barley which is the marginal source of feed energy. In a LCA approach, a tonne of rapeseed oil causes increased production of palm oil and displacement of both soy meal and a small amount of barley.

If equilibrium models aggregate all of the oilseeds in a region, it can be argued that the model cannot differentiate between the effects of marginal meal and marginal oil demands. It is also important that in the models that do disaggregate oil seeds, soybean should be modelled as the marginal source of high protein meal, thus allowing for the reduced growth of soy bean production due to substitution of soy meal by biofuel by-products. Substitution ratios for EU biofuel by-products based on balancing protein and energy requirements in compound animal feed are shown in Table 1.

Substitution Ratios for EU Biofuel co-products			
	Co-product	**Substution for soy meal**	**Substution for cereal**
		t / t of co-product	
CE Delft 2008			
	Wheat DDGS	0.50	0.66
	Maize DDGS	0.45	0.69
Lywood et al 2009			
	Wheat DDGS	0.59	0.39
	Maize DDGS	0.40	0.49
	Rape meal	0.61	0.15
ADAS 2010			
Current scenario	Wheat DDGS	0.33	
Future high usage scenario	Wheat DDGS	0.60	

Table 1: **Co-product substitution ratios [presented by W. Lywood - ENSUS]**

[29]Schmidt J H, Christensen P and Christensen T S (2009). Assessing the land use implications of biodiesel use from an LCA perspective. Journal of Land Use Science Vol. 4, No. 1–2, March 2009, pp 35–52

Whilst FAPRI and the latest versions of GTAP allow for substitution both report that there is still scope for using more DDGS to replace soybean meal or other protein feeds. It was also noted that AGLINK-COSIMO has a sophisticated disaggregated substitution regime. All models agree that biofuel production will increase the price of carbohydrate animal feed and reduce the price of protein animal feed. The net effect on meat prices is small, but generally positive.

EPA took into account by-products from the biofuel pathways modelled (in both FAPRI and FASOM models).EPA modelled how changes in by-product supplies impact relative prices in commodity, feed and food markets. The modelling also accounted for the feed values of different by-products for different types of livestock. For example, when meal prices decrease this can reduce the price of pork and poultry relative to beef.

In the JRC-IE model comparison, it was impossible to isolate the by-product effect in terms of area, so the effect was reported in terms of tonnes of production (Figure 21).

Model and scenario		Feedstock (tonnes)	By-products
FAPRI-CARD	EU Wheat Ethanol	**5.4**	31%
	EU Rapeseed Biodiesel	**3.0**	61%
GTAP	EU Wheat Ethanol	**5.2**	32%
	US Coarse grains Ethanol	**4.6**	31%
	EU Biodiesel (mix)	**2.4**	52%
	Malay_Ind Biodiesel	**5.1**	22%
LEITAP	Maize Ethanol US	**5.0**	7%
	Wheat Ethanol Fra	**5.5**	1%
	Biodiesel Deu	**3.0**	1%
	Malay_Ind Biodiesel	**3.0**	0%

Figure 21: Fraction of gross feedstock saved by by-products (tonnes) Presented by JRC-IE)

These results are roughly in line with the fraction of ILUC area saved estimated by running the models with by-products “turned off”. The version of LEITAP run for JRC-IE showed very low by-product effects, but by-products treatment in LEITAP has now been improved; they now save about 20-25% in terms of area.

The by-product effect in terms of area depends on several assumptions made in the models. Many models spread both the demand for extra land needed for cereals and the protein land credit around the world. However, if all of the wheat for biofuel is produced within the EU at an average EU wheat yield (the highest in the world) and all of the by-product benefit on reduced soybean area is felt in just one country e.g. Brazil, at developing-country yields then the area savings would be greater than reported in scenarios that spread the demand for extra land and land credit around the world.

It was claimed that the primary reason for the difference in results between the approach used by E4tech (2010) and the approach used in economic models, is that some of the economic models do not perform a protein mass balance between DDGS and the animal feeds that are displaced by DDGS.

4.3.2 ILUC reduction from less food consumption.

Reduced food consumption is an important market response to increased biofuels production (Hertel et al. (2010). Hertel et al. (2010) reported that the GTAP-Bio model shows changes in food prices and consumption for all food categories in the US and globally. When US coarse grain prices rose, there were reductions in consumption for coarse grains and many other agricultural and food products.

Direct consumption of coarse grains is only modestly affected in the US (-0.9%), owing to price-inelastic demand. Despite a smaller price rise, consumption of livestock products (more price-sensitive) falls by more. In the world as a whole, consumption of all food falls. While lower food consumption may not translate directly into nutritional deficits amongst wealthy households, any decline in consumption can have a severe impact for households that are already malnourished. When Hertel et al (2010) ran the GTAP-Bio model holding food consumption fixed with a series of country-by-commodity subsidies ILUC increased by 41%. This estimate may be thought of as a "food-neutral" ILUC value or alternatively, as an ILUC value that translates food effects into units of GHG emissions.

The Californian Air Resources Board (ARB) and Hertel et al (2010) report that the majority of models considered for the calculation of the Low Carbon Fuel Standard (LCFS) predict reductions in food (and/or feed and fibre) consumption, representing part of the process by which land use change for biofuel production increase is not a hectare-for-hectare displacement of existing area (ARB, 2010).

Keeping food consumption constant in a model does not fully illustrate the effects of how LUC with reduced food consumption because many other factors change in the model as well. JRC (2010b) calculated the net reduction in food and feed consumption reported by the several models using mass balance:

Where:
Net increase in world total crop production in tonnes (reported by model)
equals
Gross feedstock requirements for biofuels
minus
Tonnes of crops replaced by by-products
Minus
Reduction in food and feed consumption

JRC (2010b) showed that most of the models included in the comparison indicate a reduction in crops used for food and feed due to biofuels production. The percentage of food consumption reduction (after accounting for by-products) was highest for ethanol scenarios, and was generally highest in the models which reported the least ILUC (See Table 2)

Model and Type of Ethanol	*Food Consumption Reduction (exclusive of by-products)*
GTAP US Maize	**52%**
IMPACT US Maize	**36%**
IMPACT EU Wheat	**47%**
FAPRI CARD EU Wheat	**34%**
GTAP EU Wheat	**46%**

Table 2: Food consumption reduction in JRC model comparison results (after by-products have been accounted for).

Arguably, this effect should not be included in the calculation of ILUC emissions. If the effect is removed, the ILUC emissions from all models in the JRC modelling comparison would appear to outweigh the direct GHG savings for all the biofuels with the possible exception of sugar-cane ethanol (which is not included in the calculations shown below). In Table 3 the C content of the reduced food consumption in total (not just for individual crops) is used to calculate the CO_2 emissions credit from the food consumption reduction. Production emissions were calculated using JRC Well-to-Wheel study v3 values.

Model & Type of Ethanol	Production Emissions (gCO_2/MJ)	Accounting for food consumption reduction		NOT accounting for food consumption reduction	
		LUC[1] (gCO_2/MJ)	Total CO_2 emissions (gCO_2/MJ)	LUC[1] (gCO_2/MJ)	Total CO_2 emissions (gCO_2/MJ)
	JRC WTW v3	JRC-IE estimated results	Production and LUC	JRC-IE estimated results	Production and LUC
Purdue GTAP US Maize	47.2	36.6	**83.8**	88.4	**135.5**
Purdue GTAP EU Wheat[2]	49.7	140.2	**189.9**	191.9	**241.7**
IFPRI Impact US Maize	47.2	18.7	**65.9**	76.5	**123.7**
IFPRI IMPACT EU Wheat[2]	49.7	39.0	**88.8**	116.8	**166.5**
IFPRI IMPACT EU coarse grains[2]	47.2	20.3	**67.5**	38.1	**85.3**
FAPRI EU Wheat Ethanol[2]	49.7	69.0	**118.7**	127.9	**177.6**

Table 3 CO_2 emissions credit from food consumption (JRC update of figure presented by Searchinger)

1. LUC emissions estimated by JRC-IE in the modelling comparison report.
2. Average of CHP – Combined heat and power using natural in the processing and No CHP means NG boiler for process heat.

The potential of biofuels to reduce GHG emissions is based on having some other sink or some other reduced source of emissions (an offset). In effect we are trying to use plant growth to offset the emissions from biofuel production, but you can only count additional plant growth. If you were to only have a direct land use change that was beneficial, e.g. irrigation of a desert for additional plant growth, that is a direct reduction. But if you use crops that would be grown anyway, which is the premise of the ILUC calculation, there is no direct reduction. One place you can get a GHG reduction is if the food is not replaced. If you have crops, (e.g. wheat for ethanol), the wheat is not counted as C sequestration, because it is typically consumed and respired by livestock and people. And if the wheat is not replaced, which means that LUC will be less, there is a GHG reduction. But the GHG reduction physically comes from the reduced respiration and waste by people and livestock. The models are implicitly calculating this reduced consumption, which results in reduced emissions of carbon and attributing that to the benefits of the biofuel. By assuming the credit in the LCA and not counting the emission of this carbon, because of the reduced food consumption, that is what is effectively happening. By calculating the actual GHG emissions that the models are implicitly crediting from reduced food consumption and adding that to production emission the emissions exceed that of fossil fuel.
It was argued that there may be some miscounting or misunderstanding of the changes of flows:
However, in response it was stated that:
1. It is the traditional LCA, not the ILUC calculation that attributes carbon in the crop as an offset to the carbon in the emission. That is where in effect you are using the carbon flow as an offset.
2. ILUC is not merely a change in a one-time reduction in carbon stock, it is a change in carbon stocks plus the change in carbon sequestration that would occur on that land if the NPP went into that or if what you are diverting is a food product, the subsequent knock on change in both carbon sequestration and carbon stocks to replace that on-going product of NPP.

4.4 Food commodity markets

Models tend to underestimate the long-term ease of substitution of one cereal for another and one edible oil for another, because the only statistically-significant data available is based on short term annual variations. Consumers and manufacturers need time to change their habits and processes, so substitution is easier in the longer term. Similarly, models use short-term data to assess how much demand or supply shocks in one country are spread around the world. This tends to underestimate the interconnectedness of world production in the long-term: for this reason one model, IFPRI-IMAGE, prefers assuming a single world market. Most countries import most of their vegetable oils from a few big producers. Vegetable oils easily substitute each other over the longer term, and industry experts consider palm oil to be the marginal source of vegetable oil in the world, as it has a production-cost advantage and hardly competes with other crops. That means use of vegetable oils or animal fats for biodiesel will, on a decade-timescale, largely be met by increased palm oil production. Cereals clearly can also easily more easily substitute each other in the long term. In general, crop production has shifted increasingly to land-rich countries in S. America. An exception to this trend is EU, India and China which, historically, kept cereals production mostly just ahead of demand by adjustments to domestic agriculture policies and trade barriers. However, free-trade agreements (which are taken into account in economic models) limit the EU's power to do this in the future. Therefore, in the future, increased EU cereals demand is likely to be met largely by imports or reduced exports, rather than changes in EU production. One of the main reasons for building the economic models was to see how these conditions of trade will affect future markets.

Substitution of oilseeds and animal fats

In the context of estimating critical issues in the ILUC emissions, it is important to understand what happens if there will be more demand for rapeseed oil or animal fats, and what they could be replaced with. The National Renewable Energy Action Plans (NREAPs) for EU member states show that approximately 74%[30] of the biofuel target in 2020 will be biodiesel, which implies that oil and fats are an issue. The NREAPs show that the EU will need about 22 Mt of natural oils and fats by 2020, to meet the 10% target. EU production is unlikely to keep up with demand, and there will be an increasing need to import, which will have a substantial impact on the global balance of oils and fats price relationships. These oils are now closely trading in line with the petrol/oil prices, and this is not driven by supply, demand, yield increase, area or farmers knowledge, but by the market speculation.

The world's supply in oil and fats is approximately 165 Mt, and the average increase over the past years (inclusive of any European biofuel increase) is 5-7 Mt. 75% of those oils and fats are used in food nutrition, 6% in animal feed, 10% in olio chemicals, and 9% for biofuels (see presentation from Klaus Nottinger, APAG) . Half of the world's supply of oil and fats comes from palm oil and soy, so the critical issue of ILUC will continue. Area increase and yield improvements in palm oil have been significant though the area effect has been stronger than the yield effect. In contrast to the increase in palm oil production, the increase in rapeseed production has been much less significant. Indonesia is overtaking Malaysia in palm oil production mainly because Malaysian production growth on available land has been limited because of ecological reasons and other impacts.

Palm oil is a direct substitution for tallow (a co-product from meat production). Most of the palm oil produced is exported and accounts for 60% of all the World's exported oils and fats, whereas soy and rapeseed exports are declining. For biofuel legislation, as well as food demand (at least for India and China) palm oil is helping to fulfil those demands. European consumption of rapeseed oil has been driven by biofuels (67% in 2010), with the remainder for food and other uses. The limitations of rapeseed oil production in the EU are already at a certain limit and any additional demand will have to come from palm oil.

In many applications the only substitution for palm oil is tallow. The variety of applications of tallow means that the negative ILUC emission issue will catch consumer attention, because it will impact the daily lives of the people worldwide. It is often assumed that animal fat is waste or a residue and that using it as a biofuel would allow the legislation to conclude that it should have significant GHG savings. However, this product is actually in direct conflict with a vegetable oil, which will increase GHG emissions. Rather than a credit it will have a debit of at least the palm oil number plus the rendering process. In conclusion these animal by-products cannot be considered waste, as they have a ready market.

[30] NREAPS for 26 countries, excluding Hungary indicate a 74% share for biodiesel in total biofuel consumption – JRC-IE

Trade Modelling Issues modelling

Armington elasticities

Partial equilibrium models such as FAPRI-CARD include transport and logistics costs in the least-cost global solution equalizing supply and demand. General equilibrium models use Armington elasticities to describe how easily commodities are traded between countries. A high elasticity means the market tends towards a single global market, whereas a low one means that demand in one country is largely met by local production. The land use change due to EU biofuels will generally be higher for a single world market.

Differences between crops

Oilseeds and vegetable oils are heavily traded across frontiers, and can be said to approximate to a global market (Figure 22).

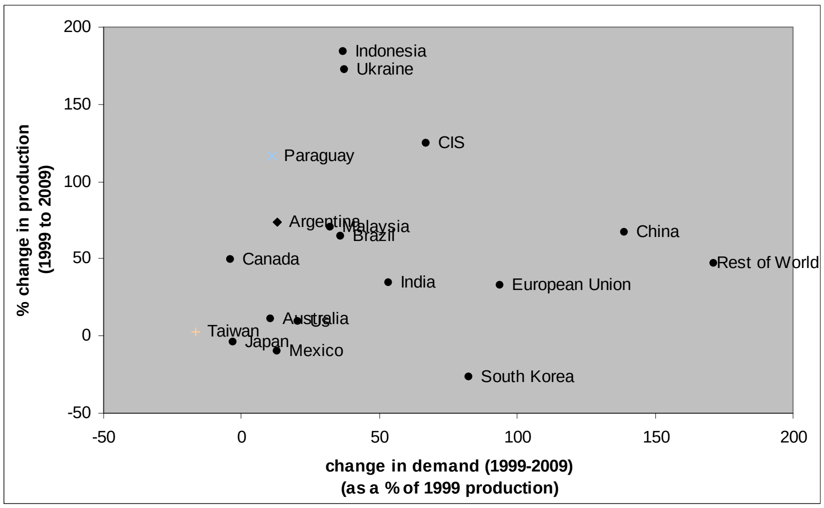


Figure 22: Demand-change and production-change (supply) of vegetable oil – Presented by JRC-IE.

However, this is less true for cereals, where increased demand in such countries as EU (e.g. Figure. 23), China and India has, historically, been met mostly by internal production.

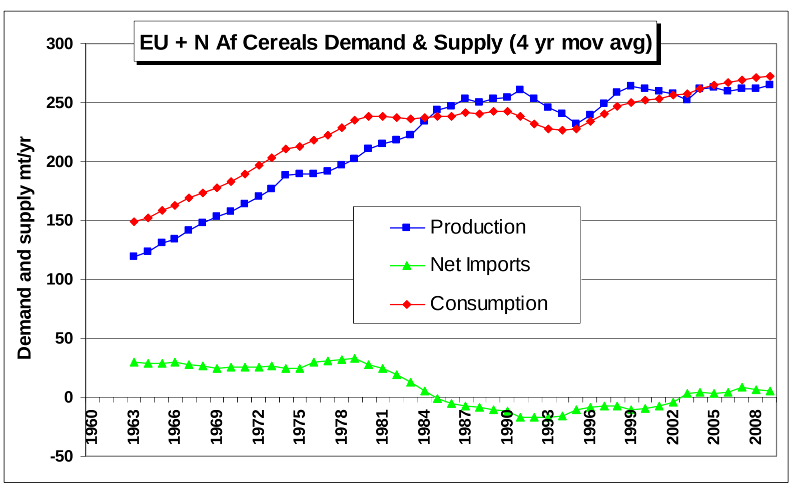


Figure 23 Demand and supply of cereals – EU and North Africa (Presented by W. Lywood - Ensus)

It was proposed that this was due to a policy of self-sufficiency, not because of market forces. The EU market, in particular, is now much more open to world trade, so history is a poor indicator of future behaviour. Furthermore, countries like Australia and Canada, which have always been open to the world market, would show the opposite conclusion: production is largely unrelated to domestic demand.

Should Armington Elasticities be differentiated between crops?

Because of these differences, it was suggested that Armington elasticities should be made crop-dependent and tuned to match historical trade data for each crop. But cereals and oilseeds are grown on the same land: thus, for example, an increase in demand for cereals in EU would be met partly by more oilseed imports. Therefore there is an argument for a single value which refers to land substitution as a whole.

Are Armington Elasticities chosen appropriately?

There are large differences between the Armington elasticities for different models: GTAP uses a value of 2.6, whereas IFPRI-MIRAGE uses 10, which seems arbitrary. However, this is because the GTAP value is for an aggregate of cereals, whereas the IFPRI value is applied to individual cereals. As individual cereals easily substitute each other, the world market can easily adjust to supply more of a particular cereal to one country. Armington elasticities are chosen to be consistent with observed price transmissions between markets: a high value is needed to explain why the price of a commodity in one country is heavily influenced by the world price.

Short-term vs. long-term data

It was proposed that modelers systematically underestimate both long-term Armington elasticities, and long-term elasticities of substitution between crops. That is because the only data available is short-term annual data, whereas the models are applied to changes over a decade. It takes time for consumers and processors to adapt to a different type of cereal or vegetable oil from a new source-country. Therefore short-term elasticities are too low in the context of a long-term model. This means that models tend to underestimate land use change in exporting countries, and to overestimate the differences in land use change between different feedstocks.

5. Sensitivity and uncertainties in modelling studies

To estimate the effects of incremental demand for biofuels with economic models, it is important to understand that there are additional "costs" that have to be considered: for instance, increased biofuels demand can be met either through a reduction in (food) consumption, with low LUC, or can be met with increased production. In the latter case it is necessary to know how much of the increase will come from LUC and how much from intensification. As shown in Figure 24 (from IFPRI analysis), for the crops selected most of the changes in production will come from LUC (green bars), and less production will come from intensification (blue bars).

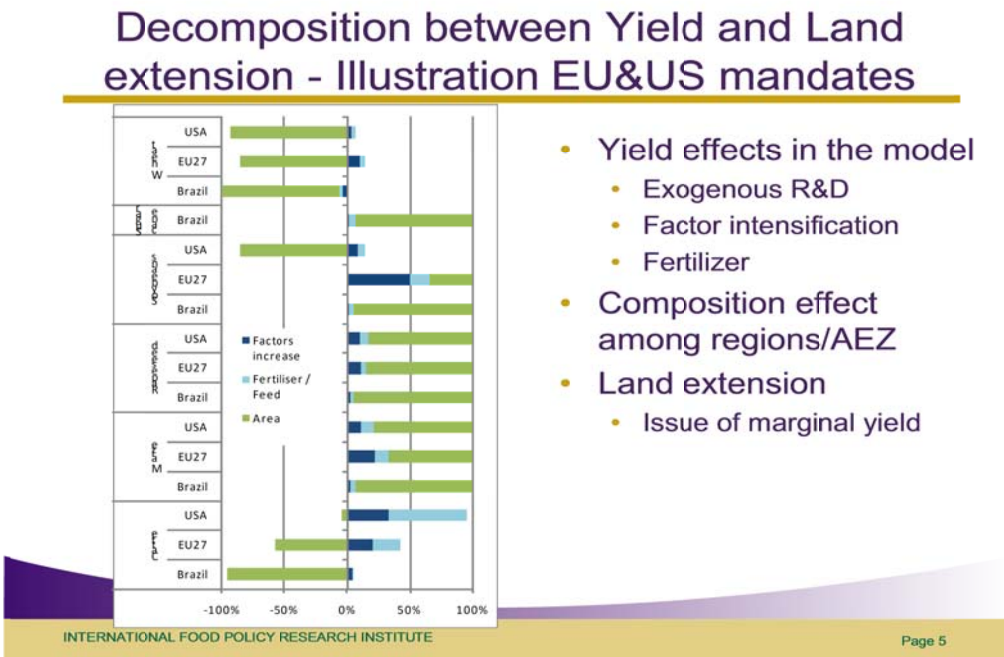


Figure 24 Decomposition between yield and land extension [presented by D. Laborde – IFPRI]

Intensification itself brings additional "costs" (more irrigation, more energy consumption, more mechanization etc.) and using LCA coefficients that are computed for just one technology ("non-intensification") does not properly account for C emissions. It is not correct in principle to assume that LUC emissions are "mitigated" through intensification, unless the above additional costs are properly accounted for. Estimating all the effects (land use change, reduction in food consumption and intensification) together will reduce errors and will match policy choices to desired outcomes better than computing only land use effects.

Another important aspect that needs to be considered is that "Operational" Global Warming Intensities (GWI) for fuels (which reflect their climate impact) that are used in policy (for example CARB values for US corn ethanol or Brazilian sugar cane ethanol) do not necessarily reflect the fuels' actual climate effects. These are two very different concepts.

There are three distinct variables for a single fuel:

1. The different models (such as IFPRI) make an estimate of the "physical" GWI of a biofuel. A pair of these estimates the GW effect of a MJ-for-MJ substitution of one fuel for another. Different models give different values for each fuel type.
2. The real "physical" GWI is not known and is a random variable, but evidence about the real GWI is provided by the models.
3. A government then decides on an 'operational' GWI of fuel to publish (e.g. in LCFS implementation)

The optimal value for this last variable–the value that produces either the least global warming, or the greatest total social value of all kinds–is not necessarily the most likely value of a fuel's physical GWI, for two reasons:
1. The uncertainty associated with the physical GWI is asymmetric, and the cost of being wrong (difference between operational and real physical GWI) may also be asymmetric. It is a conventional result of decision analysis that in these circumstances, the optimal action for a decision maker may be different from the one implied by a standard central estimator of the uncertain random variable.
2. The result of implementing a policy with a given operational GWI is not MJ-for-MJ substitution of fuels, but a more complicated response involving the relative prices of the fuels, shifting of fuels among regulated and unregulated markets, non-fuel-related regulatory changes outside the jurisdiction implementing the original policy, and various kinds of rent-seeking.
The "operational" numbers depend on the models and assumptions used to estimate the real GWI values and reflect their intrinsic uncertainties, which then cause problems in decision analysis to define an appropriate value for each fuel. The fundamental questions for the policy process are then:

- How big is the uncertainty in a given estimate and across all estimates? Importantly, this includes not only parameter uncertainty, but substantial model uncertainty, i.e. lack of knowledge about the correct functional relationships among parameters, and even which parameters to incorporate into the model.
- What is the form of this uncertainty?
- How can we deal with it in legislation and policy implementation?

Various studies are on-going: IFPRI for example is carrying out Monte Carlo simulations, with the following 7 varying parameters:

1 Elasticity on the use of inputs
2 Elasticity of substitution of land among crops
3 Elasticity between low substitutable crops
4 Elasticity of managed land expansion
5 Share of expansion in primary forest
6 Elasticity of cattle intensification
7 Marginal yields

They used three different scenarios for a 7% mandate of biofuels for transport in the EU: one based on ethanol (domestic production), one on ethanol with trade liberalization and one where all incremental biofuels comes from biodiesel. Preliminary results, obtained using a uniform distribution probability function, are shown in the Figure 25, with values for ethanol in the range of about 20 to 55 gCO_{2eq}/MJ, and for biodiesel in the approximate range 40 – 90 gCO_{2eq}/MJ.

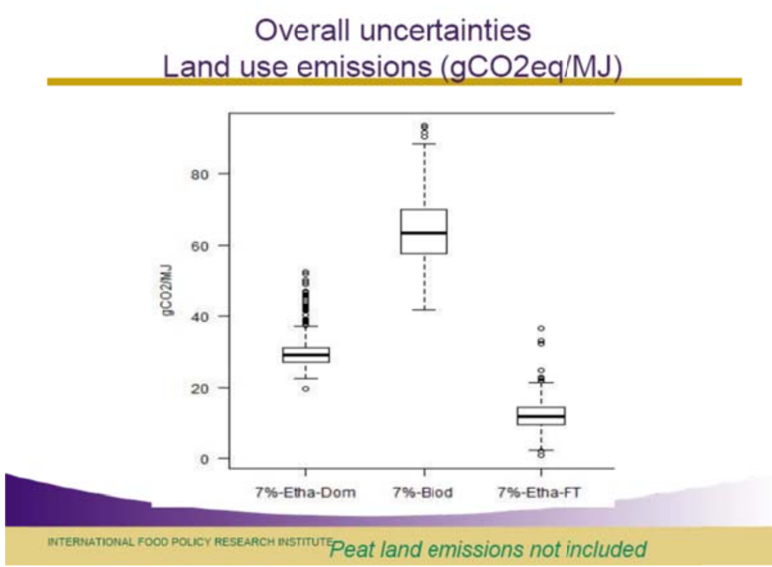


Figure 25: results of Monte Carlo analysis on Land Use emissions from IFPRI – MIRAGE model [Presented by D. Laborde – IFPRI]

A study recently published by [Plevin et al, 2010] was also presented during the meeting, showing results of the uncertainty analysis on US corn ethanol GHG emissions (Figure 26).

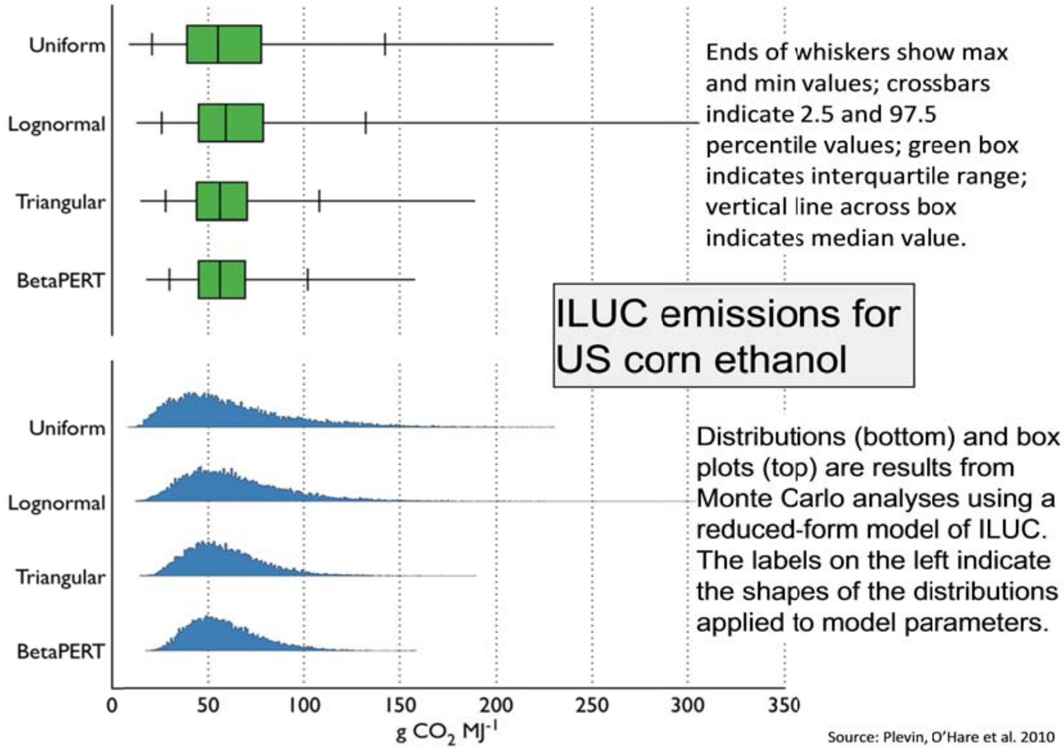


Figure 26 : Uncertainty analysis on ILUC Emissions from US corn ethanol [presented by M. O'Hare]

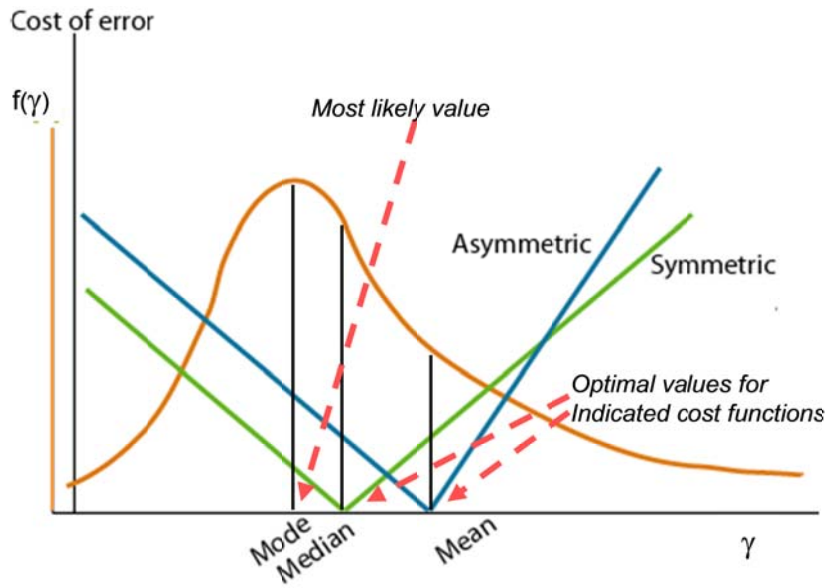
In line with the figures presented by IFPRI, this analysis evidences two important points:

- ILUC emissions are above zero, and the most likely value (the peak of the probability distribution) is significantly above zero.
- The right-side tail of all the curves indicates that higher values for ILUC emissions cannot be ruled out.

Unique aspects of this study are the inclusion of uncertainty about which type of economic model is most representative, and analysing the joint uncertainty across the combined economic and ecosystem models. Other analyses of ILUC uncertainty have considered only uncertainty in the ecosystem component (USEPA, 2010 RFS2 analysis) or in the economic model (IFPRI, 2011). The lack of certainty about the country or even continent in which ILUC occurs results in a wide plausible range for CO_2 emissions from land cover conversion, and the combined uncertainties in the magnitude of ILUC and the resulting emissions produces heavily skewed (right-tailed) distributions.

In determining 'operational' GWIs (g) for legislative purposes, it is necessary to consider both the shapes of the probability distribution function f(γ) of the real GWI (γ) (i.e. how can the "operational" GWI be determined given the shape of the probability distribution function?) as well as the cost of errors (i.e. the costs for the overall objectives of the policy of an assigned operational GWI different from the unknown real value). These costs should be evaluated considering the variety of goals of the policy as a whole (e.g. reduction of climate change impacts, energy security, development of specific branches of the economy etc.).

For any linear and symmetric (i.e. the cost of being mistaken in excess or defects is the same) cost function, the value of f(γ) which corresponds to minimum costs is the median of the distribution (as shown in the following figure). However, if the distribution function f(γ) has a long right tail towards higher values of γ as in this case, then the linear (non-symmetric) cost function is minimised a t a value higher than the mode.



5.1 ILUC and cost of carbon abatement#

At present, biofuels are generally considered to be more expensive than fossil fuels. E.g. for a biofuel achieving a 50% direct saving against an 86.4 gCO_2e/MJ fossil fuel comparator the cost for carbon abatement is $150[31] per tonne of carbon dioxide saved, ignoring iLUC (based on JRC 2008). Including ILUC emissions would raise the costs significantly. In the example shown in Figure 27, with ILUC emissions of 35gCO_2/MJ, the carbon abatement cost rises to $1000. This is of interest as often policy decisions are based on what is most cost effective.

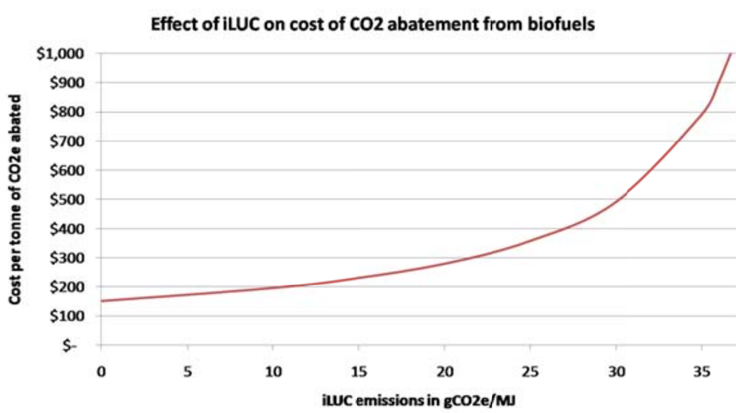


Figure 27 Abatement costs as a function of ILUC emissions (presented by C. Malins – The ICCT)

It is then clear that investing in carbon abatement measures to mitigate ILUC will bring to substantial cost abatement, and it is therefore worthwhile.

[31] or around $240 per tonne of fuel for biodiesel

5.2 Mitigating the risk of unwanted indirect effects

While it is important to know the size of the ILUC effect it is equally important to know what can be done to mitigate these indirect effects.
At the global-level preventing unwanted "direct" LUC in other sectors will also prevent the indirect effects. The pressure from the agricultural sector on land expansion as a whole can also be reduced by increasing yields, supply chain efficiencies and/or a reduction in consumption. Both of these measures are comparatively long-term solutions.
At the producer level, production of biofuel feedstocks should be expanded minimizing the risk of unwanted indirect effects. One option to do this could be to add a feedstock specific ILUC factor. This approach has the advantage that can incentivize the use of feedstocks with a low ILUC risk (e.g. sugar-cane, lignocellulosic, residues, algae etc.), but it does not provide any incentives for how these feedstocks are cultivated, and effectively excludes many feedstocks.
When ILUC emissions are added to the typical RED values for direct GHG emissions (Figure 28) it is evident that most of the feedstock would fail to meet the 50% threshold for GHG savings that will come into force in 2017.

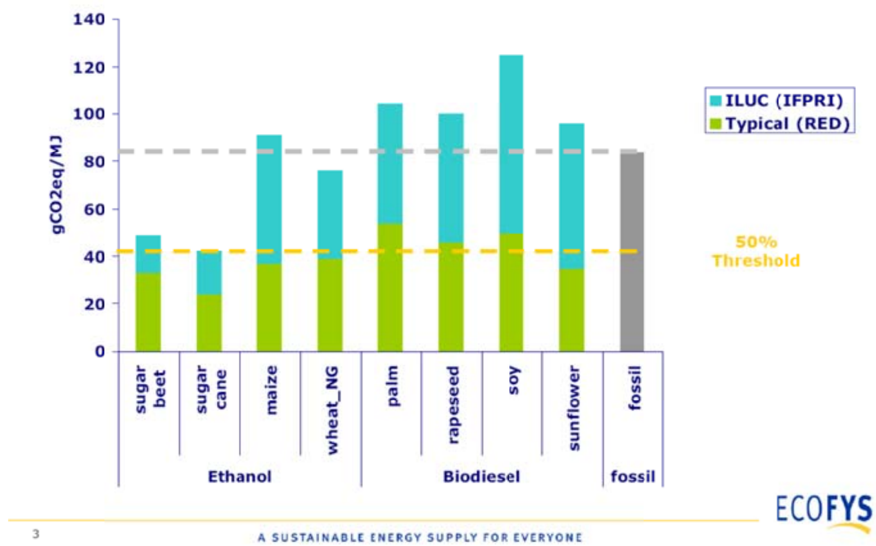


Figure 28 GHG emissions from several feedstocks compared to fossil fuel comparator. [presented by B. Dehue – Ecofys]

It is thus very important also to implement additional measures and sustainable production practices to grow feedstock without the indirect effects.
What is causing ILUC is the displacement of food production to biofuels. Therefore, if these indirect effects have to be avoided, biofuels should grow without any displacement occurring.
(N.B. This can still result in direct land use change but that problem may be more identifiable and controllable than indirect land use change).
One way this could be achieved is by expanding production on unused or underutilised land with low biodiversity and carbon stocks, also avoiding any loss in carbon sequestration (e.g. from afforestation of abandoned land). For example expansion of palm oil on Imperata grassland has great potential and appears to be economically better than expansion onto peatland. In addition the C effects of going from Imperata grassland to palm oil may be positive.
Another way to avoid displacement is to integrate the productivity of non –bioenergy systems such as cattle in Brazil with the production of sugar cane. The cattle could be fed with hydrolysed bagasse[32], thus increasing cattle density and avoiding displacement to new pasture land.
A detailed analysis of the impact of implementation of these measures on biofuels costs has still to be carried out.

[32] A small amount of bagasse is used in co-generation but this still leaves a large amount left for animal feed

It was also underlined during the discussion that better land cover data is needed to investigate alternative production systems as shown here to capture the full potential.
The risk of ILUC could also be minimized through other policy actions and supplementary measures to biofuels, for example incentivizing the use of electric cars and improving vehicle efficiency to reduce the demand for oil.
The figure below shows the results of an analysis carried out by EMPA (CH) on the potential of substitution of fossil based mobility in Switzerland. It is evident from the figure that the substitution potential of electric vehicles (from PV or cogeneration) is higher than the one of all biofuels (1st and 2nd generation).

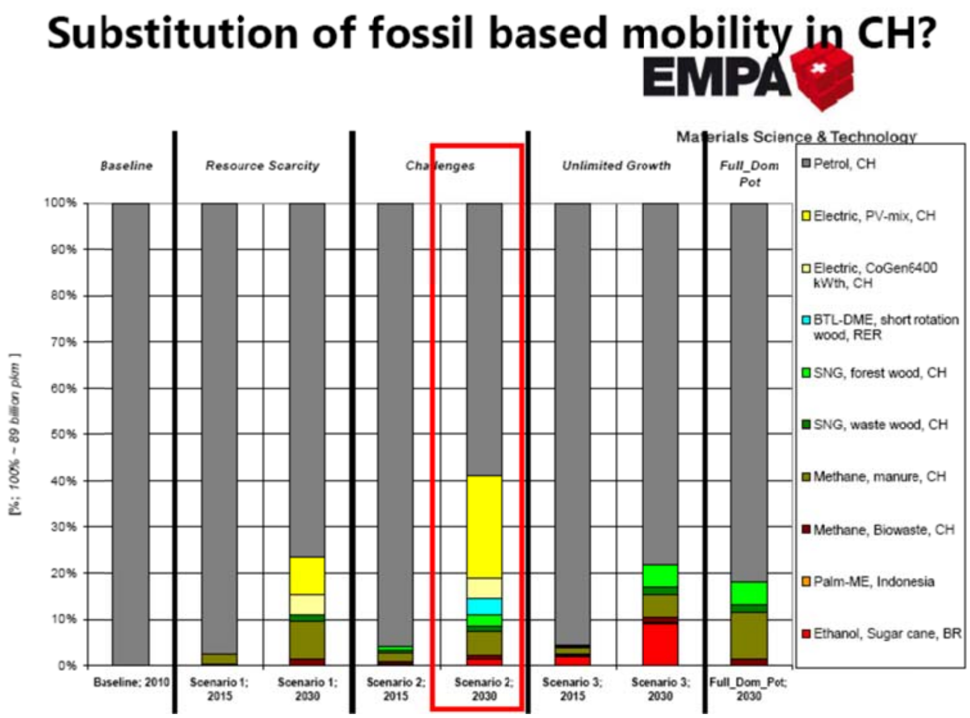


Figure 29: Fossil fuels substitution according to different scenarios (presented by J. Reinhard – EMPA)

Moreover, the improvement of vehicles efficiency has the potential to save higher GHG than biofuels, as from figure below presented by EMPA (CH).

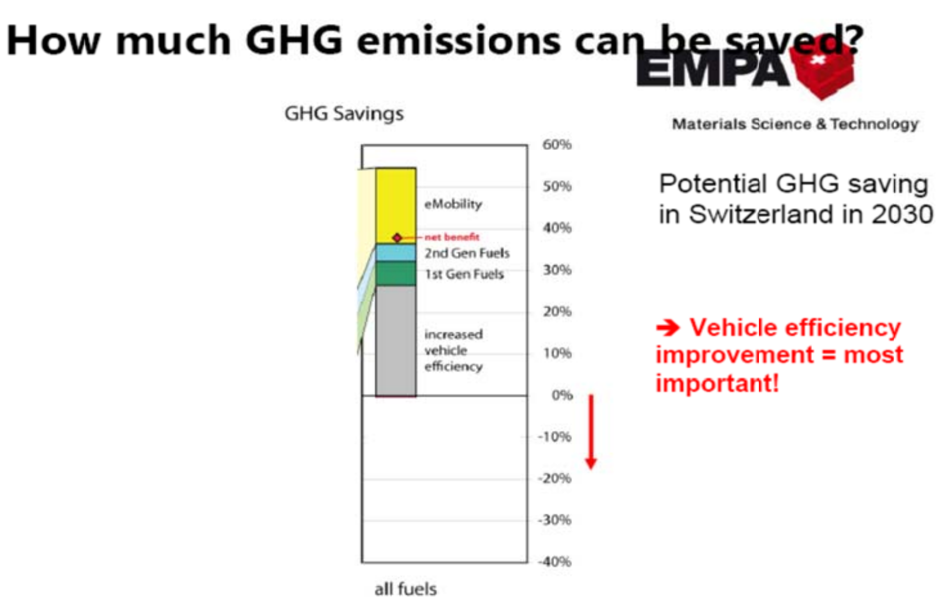


Figure 30 : GHG saving potential in Switzerland (presented by J. Reinhard – EMPA)

6. Policy recommendations on ILUC

The main objective of this section is to discuss and provide answers to the following policy-relevant questions:

Based on the available science, can we conclude that the GHG impacts from ILUC of the EU biofuel/bioliquids policy are significant?
Can we differentiate the impacts between different feedstock types, the geographical location of the feedstocks (e.g. is there a difference in the GHG impact depending on where the biofuel feedstocks are grown) and what effect do different land management/mitigation practices have.

This section includes the discussion held by the invited experts on the main policy questions (shown in bold in the following paragraphs) put forward by the Commission.

The views expressed in this report are those of the experts who participated to the workshop only, and do not represent the opinion of the Commission

Can we conclude that the GHG ILUC impacts of EU biofuel/bioliquids policy are significant?
There was consensus that:
None of the data (including uncertainties) indicate that ILUC is zero or not significant.
Most of the studies indicate that biodiesel has a higher ILUC impact then ethanol, while the demand for diesel is higher than for gasoline in the EU.

Differences in models have been largely discussed, and it was agreed that ILUC estimations are lower in some models, mainly due to:
1. Lower rate of forest conversion
2. Larger GHG benefits from a reduction in food consumption: it was agreed that model estimates of ILUC should not include a reduction of food consumption, or at least this shouldn't be attributed an "ILUC benefit" to biofuels.
3. High Armington elasticities: models which assume high values for Armington elasticities have little crop displacement outside the region of the increased demand and most apportion production to high yield areas

There are of course other reasons for differences between models (like the use of co-products, yield elasticities, marginal yields etc.), but all this has been exhaustively explained in the previous chapters.

Can we differentiate between different geographical locations?
Although there is evidence that emissions from the same feedstock (e.g. soybean) vary according to geographic location, it is quite difficult to differentiate because markets are globally connected and there are uncertainties on the identification of the region where ILUC occurs (discrimination has also to be avoided in policy).
A possibility to account for geographic differences could be to consider the productivity of lands in different geographic locations and using a "weighted factor" according to Net Primary production (NPP) of the land. For instance the weighting factor will be higher in Malaysia/Indonesia because the productivity in these regions is higher than for example in the EU (i.e. one ha in Malaysia will have double the NPP than in Europe, and therefore one ha in one year in EU will have half the ILUC than in one ha in one year Malaysia/Indonesia, but it must be also considered that the biofuel produced from EU land will also be less).

Can we recommend Land management and Mitigation practices?
Policy makers shall implement additional mitigation criteria and incentivize mitigation practices as the use of degraded land where food crops are not grown or intensification.
Concerning the use of degraded land, it was stated that cost benefit analysis and efficiency considerations favour biomass crops such as eucalyptus to store CO2 over growing biofuel feedstocks.
Encouragement of good food policies/practices that for example incentivise intensification on pasture land will certainly bring to ILUC reductions. Models assuming a large pasture intensification effect versus extensification

have also lower ILUC effects. The intensification of cattle production in Brazil (together with high yield and direct GHG saving from co-generation) results in lower ILUC from sugar cane than from other feedstocks, and this management practice should be incentivized by appropriate policies.
Increasing the yields could also mitigate ILUC effects, but the data shown by a number of studies seems to be more promising than the reality.
Additional incentives for mitigation of land use change emissions could move a biofuel above the RED threshold for GHG savings.

<u>Should the GHG threshold be increased?</u>

The majority of experts agreed that increasing the GHG threshold, whilst bringing improvements to processes (e.g. methane capture in oil palm refineries), will not be effective for ILUC reduction. The opinion of the invited experts was that a general increase of the threshold will equally penalize all biofuels, without "rewarding" those not causing ILUC (or causing reduced ILUC). There should instead be a distinction between different pathways, incentivizing the use of "low ILUC risk" biofuels.
Moreover, the increase of the GHG threshold will not benefit biodiversity or reduce competition with food etc. Other policy options (e.g. additional incentives for biofuels with low ILUC risk, obtained by use of degraded land or by specified mitigation practices, use of crop-specific "ILUC factors") would therefore be preferable in the opinion of the scientists.

<u>Should we apply an "ILUC factor"?</u>

The use of a crop-specific factor which attributes a quantity of greenhouse gas emissions to biofuels was generally recommended, as it incentivizes directly the production and use of fuels with low estimated total Carbon Impact. Thus, 2nd and 3rd generation biofuels which do not require arable land (made from residues, wastes, algae etc.) would be favoured.
An ILUC factor is the only (indirect) way to identify which biofuels displace food production and therefore cause indirect C emissions. So it also would reduce impact on food prices according to the models.
The addition of an ILUC factor to the default direct GHG emissions now in place in the Renewable Energy directive will cause a number of biofuels not to be eligible for the EU legislation. Therefore, to avoid this, the EU policy should in parallel encourage producers to address Carbon risks through better productions systems and management practices (as already discussed above).
The traceability of the feedstock used for biofuel production is fundamental in making the ILUC factor effective.
There is still an issue on how to estimate ILUC factors: as an alternative to modelling, someone argued that an attributional approach should be used, estimating total LUC and then attributing it to various drivers. ICONE reported that such an analysis was undertaken in Brazil and an ILUC factor of 8 gCO2/MJ was found for sugar cane assuming pasture intensification.

<u>Which model should be used to use calculate ILUC?</u>

Although it would be desirable to use a single reference model, it is difficult to establish which type of model is more suitable and what are the best assumptions (e.g. a partial or general equilibrium model, or to use an "Armington" approach or "world market" approach etc.). One could build a spreadsheet model incorporating the main parameters which in practice determine the model results. Then one can replicate the assumptions made in different models, verify the points of disagreement and the sensitivity of the results to these points.
But even with the above considerations, and although further improvements could be brought in the future, the available studies are sufficient to give clear indications about the importance of ILUC effects from different crops.
An alternative proposal was that the only way to solve the problem is to have a single World market model. Instead of trying to find numbers for every crop, the ILUC emissions can be calculated for protein, oil or carbohydrate and sugar. This can then be applied as a precautionary ILUC emission and give exceptions where producers can prove that this is **not** occurring (e.g. pasture intensification).

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Abstract

Under request of DG ENER and CLIMA, the JRC organised in November 2010 an expert consultation, grouping world-recognised academics and experts in the field on Indirect Land Use Change (ILUC) effects caused by increased use of biofuels. This consultation aimed at discussing the main uncertainties related to ILUC estimations and to answer to the questions addressed in the public consultation.

The two days discussions focused in particular on the following items:

1. Land use change and greenhouse gas emissions (methodologies, datasets and uncertainties to locate ILUC and calculate GHG emissions)
2. Agro-economic modelling and uncertainties
3. Policy options

The final discussions addressed policy issues, in particular:-

- Does the modelling provide a good basis for determining how significant indirect land use change is?
- Are the impacts significant?
- Can we differentiate between bioethanol/biodiesel, feedstocks, geographical areas, production methods?

The experts unanimously agreed that, even when uncertainties are high, all indicators point towards the existence of a significant ILUC effect and the magnitude of this effect is crop-specific. The sustainability criteria in the Renewable Energy and Fuel quality Directives limit Direct Land Use Change, but they are ineffective to avoid ILUC, and therefore additional policy measures are necessary.

The use of a factor which attributes a quantity of greenhouse gas emissions to crop-specific biofuels was the favourite option discussed, but it was also agreed that policies should incentivise good agricultural practices, land management C-mitigation strategies and intensification on pasture lands.

On the other hand, the experts agreed that the increase of the GHG threshold will have only a limited effect on ILUC reduction.

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